SECTION 8: APPROACH TO ESTIMATING POLLUTANT LOAD REDUCTIONS AND ENVIRONMENTAL BENEFITS

8.1 OVERVIEW

This section describes EPA's methodology for assessing the pollutant load reductions and environmental benefits of the regulatory options developed. EPA only estimated loading reductions for Option 2 and 4. We were not able to develop a methodology for estimating loading reductions attributable to the inspection and certification provisions of Options 1 and 2. As a result, no loading reductions or benefits estimates were made for Option 1, and the loading reductions and benefits estimates of Option 2 are the same as those for Option 4.

Adverse environmental impacts attributable to construction activities have been well documented and include (but are not limited to) alteration of stream flow patterns, change in river channels, and reduction in the water quality of receiving waters as a result of increased generation and transport of sediment and other pollutants. Aquatic habitats also can be damaged as a result of reduced water quality and altered hydrology. These environmental impacts can in turn cause additional environmental and economic damage by increasing the frequency and magnitude of flooding events in vulnerable areas.

Sediment from eroded soil was used as an indicator of the total pollutant load discharged from construction sites because the models available to simulate soil and sediment generation, transport and removal are widely available and recognized. Although EPA expects that there are significant loadings of other pollutants (such as phosphorus and certain metals) generated at construction sites, and therefore significant reductions attributable to the regulatory options, there was no nationally-applicable data source available to estimate these values. As a result, the benefits analysis estimates loading reductions and benefits only for sediment.

EPA used the suite of model construction sites discussed in Section 7 and documented in detail in Appendix A as the basis for calculating loads and removals. Per-state pollutant loadings were computed from a minimum of 24 construction site models (6 site size groups and 4 land uses). In most states, the variability of soils, slope, and climate resulted in 432 construction model sites that were individually defined for the state and evaluated to estimate per-site loadings and loading reductions. The computation of pollutant loadings and loading reductions accounted for the following:

- Current state erosion and sediment control and BMP requirements;
- Soil nature and the geographic distribution of soil types;
- Land slopes and flow paths on construction sites; and
- Climate and hydrology

The geographic basis for the analysis are areas created by overlapping state boundaries with the boundaries of 19 ecoregions (Omernik, 1987). Figure 8-1a and 8-1b illustrates these geographic areas for the western and eastern states, referred to as state-ecoregion areas. There are 146 state-ecoregion areas in the assessment of the 48-contiguous states. (Hawaii and Alaska are not included in the analysis because the methodology relies on data on development and soil types that were not readily available for these two states.) For each state-ecoregion area, estimates were made of the amount of annual construction acreage and the number of associated model construction sites based on NRI data (USDA, 2000). NRI data estimates developed acreage by Hydrologic Unit Code (or "HUC"). By summing the acres of developed land for all of the HUCs within a given state-ecoregion area, the total annual developed acreage within that area could be estimated. (Ecoregion boundaries used in this assessment are based on large watersheds, which are roughly equivalent to the boundaries formed by the combinations of various HUCs.)

For each model construction site within each state-ecoregion area, the sediment generation and removal was calculated under baseline conditions and under each regulatory option using the Revised Universal Soil Loss Equation (RUSLE) (USDA, 1997) and SEDCAD (Warner, 1998), reflecting existing state programs. By summing to the national level, the total sediment reduction of the regulatory options could be estimated.

Following estimation of sediment loads for each HUC under baseline and each regulatory option, sites were randomly placed within each HUC and linked to the nearest stream reach using GIS. Loads were routed to stream reaches and in-stream water quality changes from baseline were modeled using the National Water Pollution Control Assessment Model (NWPCAM). Monetized benefits were estimated using both the continuous Water Quality Index (McClelland, 1974) and water quality ladder and willingness to pay based on Carson/Mitchell (1993). The total load reductions and benefits of the regulatory options are presented in Table 8-1.

Table 8-1. Loading Reductions and Benefits of Regulatory Options

	Option 1	Option 2	Option 4
Sediment Reduction (tons/year)	0	979,896	979,896
Net Benefit Using Water Quality Ladder	0	$28,357,000	$28,357,000
Net Benefit Using Water Quality Index	0	$15,203,000	$15,203,000

Figure 8-1a. State-Ecoregions in the Western United States

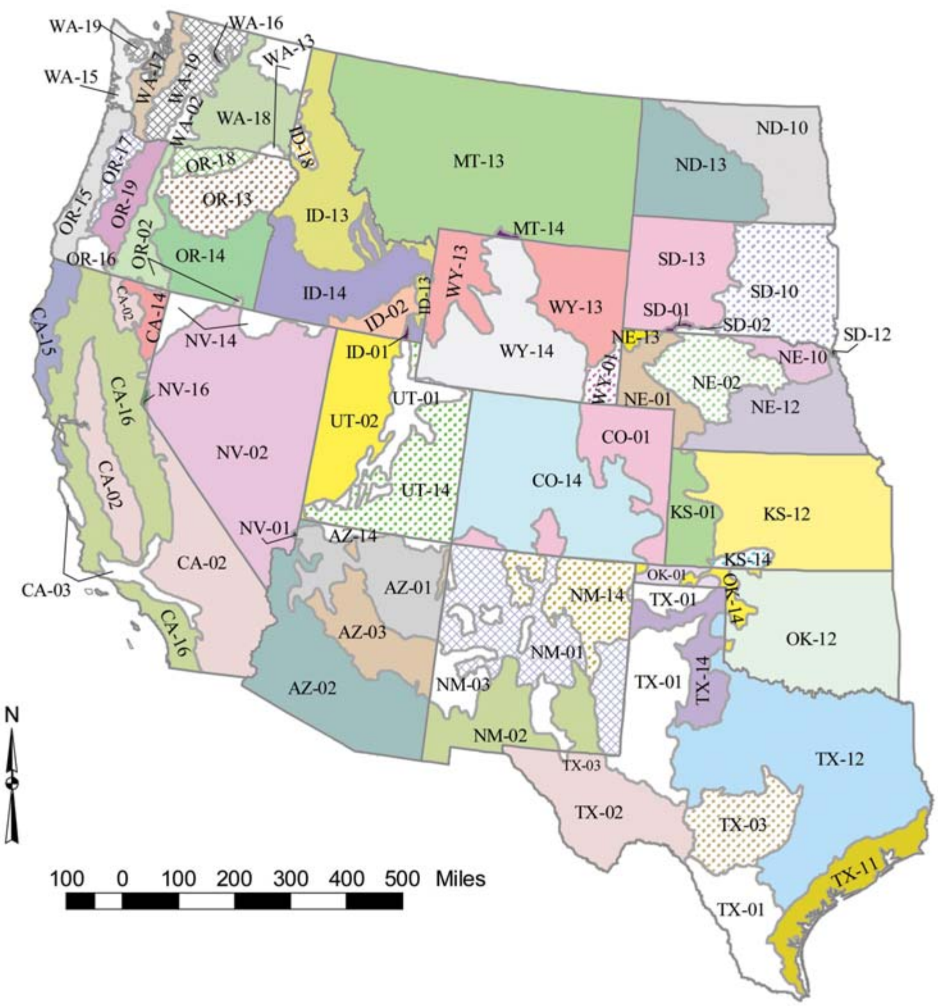
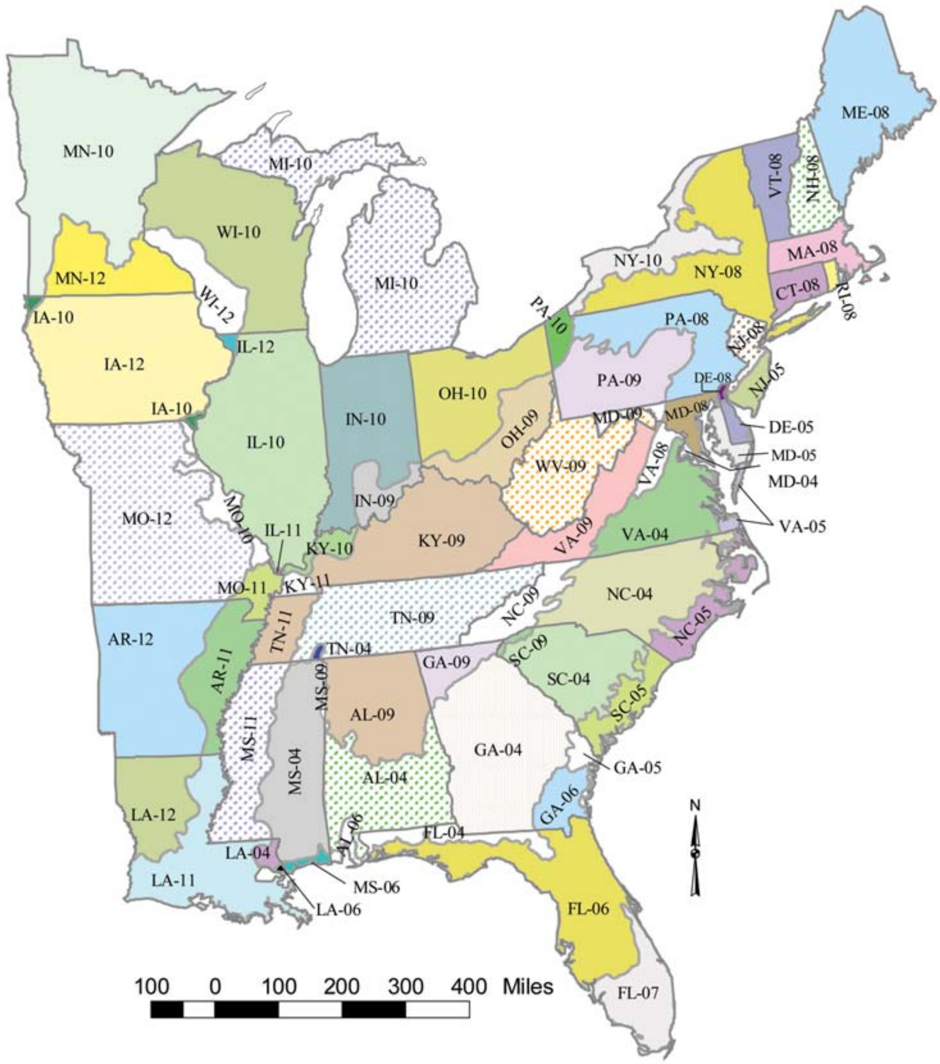


Figure 8-1b. State-Ecoregions in the Eastern United States



8.2. CATEGORIES OF REPORTED IMPACTS AND POLLUTANTS

8.2.1 INTRODUCTION

Construction and land development activities can generate a broad range of environmental impacts by introducing new sources of contamination and by altering the physical characteristics of the affected land area. In particular, these activities can result in both short- and long-term adverse impacts on surface water quality in streams, rivers, and lakes in the affected watershed by increasing the loads of various pollutants in receiving water bodies, including sediments, metals, organic compounds, pathogens, and nutrients. Ground water also can be adversely affected through diminished recharge capacity. Other potential impacts include the physical alteration of existing streams and rivers due to the excessive flow and velocity of storm water runoff.

Construction activities typically involve excavating and clearing existing vegetation. During the construction period, the affected land is usually denuded and the soil compacted, leading to increased storm water runoff and high rates of erosion. If the denuded and exposed areas contain hazardous contaminants or pollutants (either naturally occurring or due to previous land uses), they can be carried at increased rates to surrounding water bodies by storm water runoff. Although the denuded construction site is only a temporary state (usually lasting less than 6 months), the landscape is permanently altered even after the land has been restored by replanting vegetation. For example, a completed construction site typically has a greater proportion of impervious surface than the predevelopment site, leading to changes in the volume and velocity of storm water runoff. Changes in land use might also lead to new sources of pollution, such as oils and metals from motor vehicles, nutrients and pesticides from landscape maintenance, and pathogens from improperly installed or failing septic tanks. Increased pollutant loads are particularly evident when land development takes place in previously undeveloped environments. Together the short-term impacts from construction activities and the long-term impacts of development can profoundly change the environment.

Pollutants associated with construction activities and land development storm water discharges can adversely affect the environment in a number of ways. Potential effects include impairment of water quality, destruction of aquatic life habitats, and enlargement of floodplains. To the extent possible, this discussion distinguishes between environmental impacts generated during active construction and environmental impacts attributable to the more broad change in land use from undeveloped land areas such as agriculture, forest or rural area to urban conditions (termed "postdevelopment" throughout the remainder of this section). Although in most cases the differences are in magnitude and duration (e.g., sediment runoff), environmental impairment from such contaminants as pathogens are more likely to be associated with the overall urbanization of a watershed than with the types of activities that take place during construction. The discussion of environmental impacts first evaluates the impacts of contaminated runoff and then focuses on the physical impacts of construction and land development.

8.2.2 POLLUTANTS ASSOCIATED WITH CONSTRUCTION AND LAND DEVELOPMENT STORM WATER RUNOFF

There are a number of pollutants associated with construction and land development storm water runoff. This description does not represent the complete suite of contaminants that can be found in the runoff, but focuses instead on those that are currently known to be the most prevalent and of greatest concern to the environment. These pollutants include sediment, metals, polycyclic aromatic hydrocarbons (PAHs), oil, grease, and pathogens.[5]

8.2.2.1 Sediment

Sediment is an important and ubiquitous constituent in urban storm water runoff. Surface runoff and raindrops detach soil from the land surface, resulting in sediment transport into streams. Sediment level measurement can be divided into three distinct subgroups:

- Total suspended solids (TSS) are a measure of the suspended material in water. The measurement of TSS in urban storm water allows for estimation of sediment transport, which can have significant effects locally and in downstream receiving waters.

- Turbidity is a function of the suspended solids and is a measure of the ability of light to penetrate the water. Turbidity can exhibit control over biological functions, such as the ability of submerged aquatic vegetation to receive light

- Total dissolved solids are a measure of the dissolved constituents in water and are a primary indication of the purity of drinking water.

Sources of Sediment

Construction Sites

Erosion from construction sites can be a significant source of sediment pollution to nearby streams. A number of studies have shown high concentrations of TSS in uncontrolled runoff from construction sites, and results from these studies are summarized in Table 8-2. One study,

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[5] Much of the data cited in this document was collected before the NPDES Phase I and II storm water regulations generally required permits for all construction sites of one or more acres. As a result, much of this data may not accurately reflect current sediment discharge rates from construction sites. However, the data is important to reinforce the need for continued erosion and sediment control nationwide and to provide perspective on the sediment discharge rates that can occur from uncontrolled construction sites. Since even well managed construction sites continue to discharge sediment, much of the receiving water data and discussion is still likely applicable, however. This is especially true for sediment mobilized as a result of receiving channel instability following urban development, which is well documented and still largely unaddressed in many areas of the country.

conducted in 1986, calculated that construction sites are responsible for an estimated export of 80 million tons of sediment into receiving waters each year (Goldman, 1986, cited in CWP, 2000). On a unit area basis, construction sites can export sediment at 20 to 1,000 times the rate of other land uses (CWP, 2000).

Table 8-2. Studies of Uncontrolled Soil Erosion as TSS From Construction Sites

Site	Mean Inflow TSS Concentration (mg/L)	Source
Seattle, Washington	17,500	Horner, Guerdy, and Kortenhoff, 1990
SR 204	3,502	Horner, Guerdy, and Kortenhoff, 1990
Mercer Island	1,087	Horner, Guerdy, and Kortenhoff, 1990
RT1	359	Schueler and Lugbill, 1990
RT2	4,623	Schueler and Lugbill, 1990
SB1	625	Schueler and Lugbill, 1990
SB2	415	Schueler and Lugbill, 1990
SB4	2,670	Schueler and Lugbill, 1990
Pennsylvania Test Basin	9,700	Jarrett, 1996
Georgia Model	1,500 – 4,500	Sturm and Kirby, 1991
Maryland Model	1,000 – 5,000	Barfield and Clar, 1985
Uncontrolled Construction Site Runoff (MD)	4,200	York and Herb, 1978
Austin, Texas	600	Dartiguenave, ECLille, and Maidment, 1997
Hamilton County, Ohio	2,950	Islam, Taphorn, and Utrata-Halcomb, 1998
Mean TSS (mg/L)	3,681	NA

Postdevelopment Conditions

Sediment sources in urban environments include bank erosion, overland flow, runoff from exposed soils, atmospheric deposition, and dust (Table 8-3). Streets and parking lots accumulate dirt and grime from the wearing of the street surface, exhaust particulates, “blown-on” soil and organic matter, and atmospheric deposition. Lawn runoff primarily contains soil and organic matter. Source area monitoring data from Bannerman (1993), Waschbusch (2000), and Steuer (1997) are shown in Table 8-4. Hot spots (areas that are particularly high pollutant sources) were identified for the transport of sediment from the urban (developed) land surface, and they include streets, parking lots, and lawns.

Table 8-3. Sources of Sediment in Urban Areas

Source Area	Loading
Bank erosion	Up to 75 percent in California and Texas studies
Overland flow	Lawns - average value of geometric means from 4 studies: 201 mg/L
Runoff from areas with exposed soils	Average value: 3,640 mg/L
Blown-on material and organic matter	May account for as much as 35 to 50 percent in urban areas

Bannerman et al, 1993; Dartinguenave et al, 1997; Schueler, 1987; Steuer et al, 1997; Trimble, 1997; Waschbusch et al, 2000

Table 8-4. Source Area Concentrations for TSS in Urban Areas

Source Area	TSS (mg/L)[a]	TSS (mg/L)[b]	TSS (mg/L)[c] Monroe Basin	TSS (mg/L)[c] Harper Basin
Commercial parking lot	110	58	51	
High-traffic street	226	232	65	
Medium-traffic street	305	326	51	
Low-traffic street	175	662	68	69
Commercial rooftop	24	15	18	
Residential rooftop	36	27	15	17
Residential driveway	157	173		34
Residential lawn	262	397	59	122

[a] Steuer et al, 1997.
[b] Bannerman et al, 1993.
[c] Waschbusch et al, 2000.

Parking lots and streets are responsible not only for high concentrations of sediment but also for high runoff volumes. Normally about 90 percent of the water that falls on pavement is converted to surface runoff, whereas roughly 5 to15 percent of the water that falls on lawns is converted to surface runoff (Schueler, 1987). The source load and management model (SLAMM; Pitt and Voorhes, 1989) evaluates runoff volume and concentrations of pollutants from different urban land uses and predicts loads to the stream. When used in the Wisconsin and Michigan subwatersheds, the model estimated that parking lots and streets were responsible for more than 70 percent of the TSS delivered to the stream (Steuer, 1997; Waschbusch et al, 2000). Because basin water quality measurements were taken at pipe outfalls, bank erosion was not accounted for in the studies.

Sediment load is due to erosion caused by an increased magnitude and frequency of flows brought on by urbanization (Allen and Narramore, 1985; Booth, 1990; Hammer, 1972; Leopold, 1968). Streambank studies by Dartinguenave et al (1997) and Trimble (1997) determined that streambanks are large contributors of sediment in urban streams. Trimble (1997) used direct measurements of stream cross sections, sediment aggradation, and suspended sediment to determine that roughly 66.7 percent of the sediment load in San Diego Creek was a result of bank erosion. Dartiguenave et al (1997) used a GIS- based model developed in Austin, Texas, to determine the effects of stream channel erosion on sediment loads. By effectively modeling the pollutant loads on the land surface and by monitoring the actual in-stream loads at U. S. Geological Survey (USGS) gauging stations, they were able to determine that over 75 percent of the sediment load came from the streambanks.

Receiving Water Impacts

Sediment transport and turbidity can affect habitat, water quality, temperature, and pollutant transport, and can cause sedimentation in downstream receiving waters (Table 8-5). A large body of scientific literature addresses the question of how the health of aquatic resources is impacted by excess sediment loading in waterbodies. At least partly on the basis of the findings of this research, some states across the country have already set sediment targets for receiving waters to protect aquatic resources, and are developing and refining targets for geographically specific watersheds. Demarcation by waterbody type provides context and is an important theme in the literature for purposes of setting sediment targets. Differences among receiving waters are evident not only in the aquatic species that inhabit them, but also in terms of behavior of sediment within the waterbody and threshold levels of impacts. The biota or aquatic species that are the focus of the literature include aquatic vegetation, macroinvertebrates, eggs, fry, juvenile, and adult fish, shellfish and corals. Identified waterbody types in the literature include:

- lakes, reservoirs, ponds, and impoundments
- rivers and streams
- wetlands
- oceans, estuaries, and other coastal water ecosystems, including coral reefs

The impacts of excess sediment in the water include direct physical effects such as reducing visibility and light in the water column, physical abrasion of plant surfaces, clogging gill openings, and entombing of eggs and fry in redds. Impacts may also be indirect, as in changes to the chemical composition of the water, light penetration or turbidity, and/or temperature profile, which in turn affect primary productivity with repercussions in terms of fish behavior, and overall community profiles and trophic structure. Thus the aquatic resources may be directly affected in terms of aesthetics, physiology, and mortality, or affected indirectly via changes in the habitat structure of the waterbody. Bedded sediments, though they directly affect the survival of fish eggs and fry and other organisms, do so because they alter the habitat structure and are dealt with in Section 8.2.3 under Physical Impacts of Construction and Land Development Activities.

Table 8-5. Sediment Impacts on Receiving Waters

Resource Affected	Impacts of Sediment	References
Streams	• Loss of sensitive species and a decrease in fish and macroinvertebrate communities • Clogging of gills and loss of habitat • Decreased flow capacity in streams • Interference with water quality processes • Affects transport of contaminants	Kundell and Rasmussen, 1995 Leopold, 1973 Barrett and Molina, 1998 MacRae and Marsalek, 1992
Wetlands	• Deposition of sediment • Loss of sensitive species–amphibians, plants	Horner et al, 1997 Hilgartner, 1986 Pasternack, 1998
Reservoirs	• Turbidity results in increased costs of treatment for drinking water • Sedimentation results in decreased storage	Holmes, 1998
Beaches	• Turbidity reduces aesthetic value • Sedimentation can result in increased accretion rates in wetlands and change plant community structure	Kundell and Rasmussen, 1995
Estuaries	• Sedimentation • Turbidity accentuates eutrophication • Loss of submerged aquatic vegetation (SAV) • Reduced light attenuation	Pasternack, 1998 Livingston, 1996 Schiff, 1996 Mackiernan et al, 1996 Short and Wyllie-Echeverria, 1996 Orth and Moore, 1983 Stevenson et al, 1993 Hilgartner, 1986

Storm water discharges generated during construction activities cause a wide variety of physical, chemical, and biological water quality impacts. The interconnected process of erosion, sediment transport, and delivery is the primary pathway for introducing pollutants such as excess sedimentation, total suspended solids, nutrients, metals, and organic compounds to aquatic systems (Novotny and Chesters 1989) in USEPA (1999). USDA (1989) estimated that 80 percent of the phosphorus and 73 percent of the Kjeldahl nitrogen are directly associated with eroded sediment (cited in Fennessey and Jarrett (1994), in USEPA 1999). The 2000 National Water Quality Inventory (USEPA) states that siltation is one of the top causes of impairment of waters across the United States. The report also states that pollution from urban and agricultural land transported by precipitation and runoff, and which includes pollutants from construction and land development activities, is the leading sources of impairment.

Large amounts of fine sediment, or the introduction of coarse sediment is a also concern because of the of filling lakes and reservoirs and clogging of stream channels (Paterson et al, 1993, in USEPA, 1999).

The literature reviewed for this document focuses on study methodologies that describe quantitative effects of sediment imbalance in aquatic systems in a basic dose-response relationship and where aquatic organisms are exposed to suspended and/or bedded sediments. The review considered literature on each type of aquatic resource: aquatic vegetation and primary production, invertebrates, juvenile fish, fry, and eggs, and adult fish. These aquatic biota are considered within their geographical setting and waterbody type: rivers/streams, ponds/lakes, estuaries/coastal environments. Areas that are covered more extensively in the literature than other topics are the impacts of suspended sediment on adult fish and impacts of deposited or substrate sediment on juvenile fish, fry, and eggs. Cold-water salmonid fish, predominantly in a stream setting, dominates the literature on this sediment dose-response relationship. The literature is not as extensive or as rich, on estuaries, lakes, and coastal areas nor on macroinvertebrates and aquatic plants, in comparison to fish. Additional summary of biological impacts of sediment on aquatic ecosystems is available as part of the materials created as part of EPA's work on developing water quality criteria for sediments (USEPA, 2003).

Measures of suspended sediment include turbidity and total suspended solids, already covered in Section 8.2.2.1. With respect to reviewing these dose-response studies authors typically consider how either turbidity or TSS affects biota. However, the relationship between the two measures is often unclear and not explicitly defined. Turbidity is a measure of light dispersion whereas TSS measures the mass of particles in the water column. Larger particles contribute mass to a TSS measurement, but do not scatter light as much as a similar weight of smaller particles. Usually when the sediment particles are smaller, turbidity levels are higher. Suspended sediment and its resulting turbidity can reduce light for submerged aquatic vegetation. In addition, deposited sediment can cover and suffocate benthic organisms like clams and mussels, cover habitat for substrate-oriented species in urban streams, and reduce storage in reservoirs. Pollutants such as hydrocarbons and metals tend to bind to sediment and are transported with storm flow (Crunkilton et al, 1996; Novotny and Chesters, 1989). Increased turbidity also can cause stream warming by reflecting radiant energy (Kundell and Rasmussen, 1995).

Studies involving an analysis of the relationship between the two measures of suspended sediment include Packman et al (1999) who showed that TSS and turbidity have a strong positive relationship in nine urban/suburban Puget lowland streams. New Mexico TMDLs (NMED, 2002) converted a turbidity standard to TSS by calibrating with local data, so that the TSS values in units of mg/L could be converted to sediment loads in lbs/day. Keyes and Radcliff (2002) calibrated turbidity units (NTU) to approximate TSS measures using 40 mg/L kaolin clay set to a standard of 40 NTU. However, in natural streams the composition of suspended particles is not uniformly like that of kaolin clay.

The impact of suspended sediment depends on the type of particle sizes to some extent, and therefore TSS and turbidity measures should be considered together where the information is available. For example, Servizi and Martens (1992) reported that salmonids were relatively tolerant of elevated TSS levels when the particle sizes were larger. When the particles are smaller, turbidity is higher, which appears to make conditions more difficult for salmonids.

The effects of sediment deposition from construction activities are known to affect streams far downstream of construction sites. For example, Fox (1974), in USEPA (1999), found that streams between 4.8 and 5.6 miles downstream of construction sites in the Patuxent River watershed were impacted by sediment inputs. Erosion from construction sites can also generate the transport of pollutants associated with onsite wastes. The Storm Water Quality Task Force (1993), in USEPA (1999), states that rain splash, rills, and sheetwash encourage the detachment and transport of pollutants (including both sediments and pollutants associated with sediments) to waterbodies. Erosion from construction sites and runoff in developed areas can elevate pollutant loads well above those in undisturbed watersheds. Novotny and Olem (1994), in USEPA (1999), state that erosion rates from construction sites are much greater than from any other land use. The results from field studies and erosion models conducted by USDA (1970), in USEPA (1999), found that erosion rates from construction sites are usually an order or magnitude higher than row crops and several orders of magnitude higher than rates from well-vegetated areas such as forests or pastures. A review of the efficiency of sediment basins conducted by Brown (1997), in USEPA (1999), found that inflows from 12 construction sites had a mean TSS concentration of about 4,500 mg/L. Kuo (1976), in USEPA (1999), found that suspended sediment concentrations from housing construction sites in Virginia were measured at 500-3,000 mg/L, or about 40 times larger than the concentrations in runoff from already-developed urban areas. In Wisconsin, Daniel et al (1979) (in USEPA 1999) monitored storm water runoff from three residential construction sites and found that annual sediment yields were more than 19 times the yields from agricultural areas. Daniel et al identified total storm water runoff followed by peak storm water runoff as the most influential factors controlling the sediment loadings from residential construction sites, and also found that suspended sediment concentrations were 15,000-20,000 mg/L in moderate storm events and up to 60,000 mg/L in larger events. Lastly, Wolman and Schick (1967), in USEPA (1999), studied impacts of development on fluvial systems in Maryland, and found that sediment yields in areas undergoing construction were 1.5 to as much as 75 times greater than detected in natural or agricultural catchments.

The effects of road construction on erosion rates and sediment yields were also examined. In West Virginia, a road construction project studied by Downs and Appel (1986) disturbed only 4.2 percent of a 4.72 square mile basin, but it resulted in a three fold increase in suspended sediment yields. During the largest storm event, it was estimated that 80 percent of the sediment in the stream was attributed to the construction site. Hainly (1980) evaluated the effect of 290 acres of highway construction on watersheds which ranged in size from 5 to 38 square miles. He found that even in the smallest watershed, the estimated sediment yield from the construction area was 37 tons per acre during a two-year period. In Hawaii, Hill (1996) found that highway construction increased suspended sediment loads by 56 to 76 percent in basins of 1 to 4 square miles. Yorke and Herb (1978), in a long term study of subbasins in Maryland portions of the Anacostia River, found that average annual suspended sediment yields for construction sites ranged from 7 to 100 tons per acre.

Studies have indicated that the water quality impact from small construction sites may be the same or greater than large construction sites on a per acre basis. The concentration of pollutants in runoff from small sites is similar to those in large sites. In urban areas the proportion of sediment that makes it to surface waters may be the same because the runoff is delivered directly to storm drain networks, with no opportunity for pollutants to be filtered out (USEPA, 1999). MacDonald (1997), in USEPA (1999), states that storm water regulations are more likely to require controls for large sites than smaller sites. The smaller sites that lack sediment and erosion controls would contribute a disproportionate amount of total sediment from construction activities.

To test the theory that small sites have sediment loads on a per acre basis similar to large sites, the EPA gave a grant to Dane County, Wisconsin Land Conservation Department, in cooperation with USGS, to evaluate sediment runoff. In this study by Owens et al (1999), in USEPA (1999), a 0.34 acre residential development and a 1.72 acre commercial office development were evaluated. At the residential site, total solids concentrations were 642 mg/L, 2,788 mg/L, and 132mg/L for preconstruction, active construction, and post-construction, respectively. This equaled 7.4 lbs preconstruction, 35 lbs during construction, and 0.6 lbs post-construction on a pollutant load basis. At the commercial site, Owens et al found that total solids during preconstruction were 138 mg/L and 200 mg/L during post-construction, but was 15,000 mg/l during the active construction period. This equaled 0.3 lbs preconstruction, 490 lbs during construction, and 13.4 lbs after construction on a pollutant load basis. The total solids from the commercial site were similar to those in a study by Downs and Appel (1986), who evaluated the effects of highway construction in West Virginia. They found that a small storm event yielded a sediment concentration of 7,520 mg/L.

Several studies have also evaluated the total amount of disturbed land for small and large construction sites. Brown and Caraco surveyed 219 jurisdictions to assess sediment and erosion control programs. They found that of the 70 respondents, in 27 cases more than three-fourths of the permits were for sites less than 5 acres, and in another 18 cases, more than half of the permits were for sites less than 5 acres. MacDonald (1997), in USEPA (1999), evaluated data on the 3,831 construction site permits for North Carolina from 1994 through 1996. He found that nearly 61 percent of the sites 1.0 acre or larger were between 1.0 and 4.9 acres in size. Given the high erosion rates, small construction sites can produce significant water quality impairment, particularly in small watersheds. Paterson (1994), in USEPA (1999), summarized that, given the critical importance of field implementation of erosion and sediment control programs, much more focus should be given to plan implementation.

8.2.2.2 Metals

Many toxic metals can be found in urban storm water, although only metals such as zinc, copper, lead, cadmium, and chromium have been indicated as being of primary concern because of their prevalence and potential for environmental harm. These metals are generated by motor vehicle exhaust, the weathering of buildings, the burning of fossil fuels, atmospheric deposition, and other common urban activities.

Metals can bioaccumulate in stream environments, resulting in plant growth inhibition and adverse health effects on bottom-dwelling organisms (Masterson and Bannerman, 1995). Generally the concentrations found in urban storm water are not high enough for acute toxicity (Field and Pitt, 1990). Rather, it is the cumulative effect of the concentration of these metals over time and the buildup in the sediment and animal tissue that are of greater concern.

Sources of Metal Runoff

Construction Sites

Construction sites are not thought to be important sources of metals contamination. Runoff from such sites could have high metals contents if the soil is already contaminated. Construction activities alone do not usually result in metals contamination, although there is little data available on this subject.

Postdevelopment Conditions

Postdevelopment conditions create significant sources of metal runoff in the urban environment, including streets, parking lots, and rooftops. Table 8-6 summarizes the major sources of metal runoff by metal type. Copper can be found in high concentrations on urban streets as a result of the wear of brake pads that contain copper. A study in Santa Clara, California, estimated that 50 percent of the copper released is from brake pads (Woodward-Clyde, 1992). Sources of lead include atmospheric deposition and diesel fuel, which are found consistently on streets and rooftops. Zinc in urban environments is a result of the wear of automobile tires (an estimated 60 percent of the total zinc in the Santa Clara study), paints, and the weathering of galvanized gutters and downspouts. Source area concentrations estimated by researchers in Wisconsin and Michigan are presented in Table 8-7. Actual concentrations vary considerably, and high-concentration source areas vary from study to study. A study using SLAMM for an urban watershed in Michigan estimated that most of the zinc, copper, and cadmium was a result of runoff from urban parking lots, driveways, and residential streets (Steuer, 1997).

Receiving Water Impacts of Metals

Downstream effects of metals transported to receiving waters, such as lakes and estuaries, have been studied extensively. Selected studies on metal impacts on receiving waters are summarized in Table 8-8. Although evidence exists for the buildup of metals in deposited sediments in receiving waters and for bioaccumulation in aquatic species (Bay et al, 2000; Livingston, 1996), specific effects of these concentrations on submerged aquatic vegetation and other biota are not well understood.

Table 8-6. Metal Sources and Hot Spots in Urban Areas

Metal	Sources	Hot Spots
Zinc	Tires, fuel combustion, galvanized pipes and gutters, road salts *Estimate of 60% from tires*[a]	Parking lots, rooftops, and streets
Copper	Auto brake linings, pipes and fittings, algacides, and electroplating *Estimate of 50% from brake pads*[a]	Parking lots, commercial roofs, and streets
Lead	Diesel fuel, paints, and stains	Parking lots, rooftops, and streets
Cadmium	Component of motor oil; corrodes from alloys and plated surfaces	Parking lots, rooftops, and streets
Chromium	Found in exterior paints; corrodes from alloys and plated surfaces	More frequently found in industrial and commercial runoff

[a] Woodward-Clyde, 1992 (Santa Clara, CA, study)
Sources: Barr, 1997; Bannerman, et al, 1993; Steuer, 1997

Table 8-7. Metal Source Area Concentrations in Urban Areas (in ug/L)

Source Area	Diss. Zinc	Total Zinc	Diss. Copper	Diss. Copper	Total Copper	Diss. Lead	Diss. Lead	Total Lead	Total Lead	Total Lead
Citation	(a)	(b)	(a)	(b)	(b)	(a)	(c)	(a)	(c)	(b)
Commercial parking lot	64	178	10.7	9	15			40		22
High-traffic street	73	508	11.2	18	46	2.1	1.7	37	25	50
Medium-traffic street	44	339	7.3	24	56	1.5	1.9	29	46	55
Low-traffic street	24	220	7.5	9	24	1.5	0.5	21	10	33
Commercial rooftop	263	330	17.8	6	9	20		48		9
Residential rooftop	188	149	6.6	10	15	4.4		25		21
Residential driveway	27	107	11.8	9	17	2.3		52		17
Residential lawn	na	59	na	13	13	na		na		na
Basin outlet	23	203	7.0	5	16	2.4		49		32

na : not available
Sources: (a) Steuer 1997; (b) Bannerman 1993; (c) Waschbusch, 1996, cited in Steuer, 1997

Table 8-8. Metals Impacts on Receiving Waters

Resource Affected	Impacts of Metals	Evidence and References
Streams	• Chronic toxicity due to in-stream concentrations and accumulation in sediment • Bioaccumulation in aquatic species • Acute toxicity at certain concentrations	• Chronic toxicity increased during longer-duration studies, i.e. 7/14/21-day studies (Crunkilton, 1996) • Delayed toxicity (Ellis, 1986/1987) • Baseflow toxicity (Mederios, 1983) • Resuspension of metals during storms accounting for some toxicological effects (Heaney and Huber, 1978) • Bioaccumulation in crayfish (Masterson & Bannerman, 1994)
Reservoirs/ Lakes	• Accumulation of metals in sediment	• Bioaccumulation levels in bottom-feeding fish were found to be influenced by the metal levels of the bottom sediments of storm water ponds (Campbell, 1995-CWP).
Estuaries	• Accumulation of metals in sediment • Loss of SAV	• Tampa Bay (Livingston, 1996) • San Diego (Schiff 1996) • SAV losses in northeast San Francisco Bay (Orth and Moore, 1983

8.2.2.3 PAHs, and Oil and Grease

Petroleum-based substances such as oil and grease and polycyclic aromatic hydrocarbons (PAHs) are found frequently in urban storm water. Many constituents of PAHs and oil and grease, such as pyrene and benzo[b]fluoranthene, are carcinogens and toxic to downstream biota (Menzie-Cura and Assoc., 1995). Oil and grease and PAHs normally travel attached to sediment and organic carbon. Downstream accumulation of these pollutants in the sediments of receiving waters such as streams, lakes, and estuaries is of concern.

Sources of PAHs, and Oil and Grease

Construction sites

Construction activities during site development are not believed to be major contributors of these contaminants to storm water runoff. Improper operation and maintenance of construction equipment at construction sites, as well as poor housekeeping practices (e.g., improper storage of oil and gasoline products), could lead to leakage or spillage of products that contain hydrocarbons, but these incidents would likely be small in magnitude and managed before off-site contamination could occur.

Postdevelopment Conditions

In most storm water runoff, concentrations of PAHs and oil and grease are typically below 5 mg/L, but concentrations tend to increase in commercial and industrial areas. Hot spots for these pollutants in the urban environment include gas stations, commuter parking lots, convenience stores, residential parking areas, and streets (Schueler, 1994). Schueler and Shepp (1993) found concentrations of pollutants in oil/grit separators in the Washington Metropolitan area and determined that gas stations had significantly higher concentrations of hydrocarbons and a greater presence of toxic compounds than streets and residential parking lots. A study of source areas in an urban watershed in Michigan (which excluded gas stations) showed that high concentrations from commercial parking lots contributed 64 percent of the estimated hydrocarbon loads (Steuer et al, 1997).

Receiving Waters Impacts

Toxicological effects from PAHs and oil and grease are assumed to be reduced by their attachment to sediment (lessened availability) and by photodegradation (Schueler, 1994). Evidence of possible impacts on the metabolic health of organisms exposed to PAHs and of bioaccumulation in streams and other receiving waters does not exist (Masterson and Bannerman, 1994; MacCoy and Black, 1998); however, crayfish from Lincoln Creek, analyzed in the Masterson and Bannerman study, had a PAH concentration of 360 micrograms per kilogram–much higher than the concentration known to be carcinogenic. The crayfish in the control stream did not have detectable levels of PAHs. Known effects of PAHs on receiving waters are summarized in Table 8-9. Long-term effects of PAHs in sediments of receiving waters require additional study.

Table 8-9. Effects of PAHs and Oil and Grease on Receiving Waters

Resource Affected	Impacts of PAHs and Oil and Grease	Citations
Streams	• Possible chronic toxicity due to in-stream concentrations and accumulation in sediment • Bioaccumulation in aquatic species • Acute toxicity at certain concentrations	• Bioaccumulation in crayfish tissue studies (Masterson and Bannerman, 1994) • Potential metabolic costs to organisms (Crunkilton et al, 1996) • Delayed toxicity (Ellis, 1986/1987) • Baseflow toxicity (Mederios, 1983)
Reservoirs	• Accumulation of PAHs in sediment	• Sediment contamination may result in a decrease in benthic diversity and transfer of PAHs to fish tissue (Schueler, 2000) • Elevated levels of PAHs found in pond muck layer (Gavens et al, 1982-CWP)
Estuaries	• Accumulation of PAHs in sediment • Potential loss of SAV • Accumulation of PAHs in fish and shellfish tissue	• Tampa Bay (Livingston, 1996) • San Francisco Bay (Schiff, 1996)

8.2.2.4 Pathogens

Microbes are commonly found in urban storm water. Although not all microbes are harmful, several species such as the pathogens *Cryptosporidium* and *Giardia* can directly cause diseases in humans. The presence of bacteria such as fecal coliform bacteria, fecal streptococci, and *Escherichia coli* indicates a potential health risk. High levels of these bacteria may result in beach closings, restrictions on shellfish harvest, and increased treatment for drinking water to decrease the risk of human health problems.

Sources of Pathogens

Construction sites

Construction site activities are not believed to be major contributors to pathogen contamination of surface waters. The only potential known source of pathogens from construction sites are portable septic tanks used by construction workers. These systems, however, are typically self-contained and are not connected to the land surface. Any leaks from them would likely be identified and addressed quickly.

Postdevelopment Conditions

Coliform sources include pets, humans, and wild animals. Source areas in the urban environment for direct runoff include lawns, driveways, and streets. Dogs have high concentrations of coliform bacteria in their feces and have a tendency to defecate in close proximity to impervious surfaces (Schueler, 1999). Many wildlife species also have been found to contribute to high fecal concentrations. Essentially, any species that is present in significant numbers in a watershed is a potential pathogen source. Source identification studies, using methods such as DNA fingerprinting, have attributed high coliform levels to such species as rats in urban areas, ducks and geese in storm water ponds, dogs, and even raccoons (Blankenship, 1996; Lim and Oliveri, 1982; Pitt et al, 1988; Samadapour and Checkowitz, 1998).

Indirect surface storm water runoff sources include leaking septic systems, illicit discharges, sanitary sewer overflows (SSOs), and combined sewer overflows (CSOs). These sources have the potential to deliver high concentrations of coliforms to receiving waters. Illicit connections from businesses and homes to the storm drainage system can discharge sewage or washwater into receiving waters. Leaking septic systems are estimated to constitute 10 to 40 percent of all systems. Inspection is the best way to determine whether a system is failing (Schueler, 1999).

There is also evidence that these bacteria can survive and reproduce in stream sediments and in storm sewers. During a storm event, they are resuspended and add to the in-stream bacteria load. Source area studies reported that end-of-pipe concentrations were an order of magnitude higher than any source area on the land surface; therefore, it is likely that the storm sewer system itself acts as a source (Bannerman, 1993; Steuer et al, 1997). Resuspension of fecal coliform bacteria from fine stream sediments during storm events has been reported in New Mexico (NMSWQB, 1999). The sediments in the storm sewer system and in streams may be significant contributors to the fecal coliform load. This area of research certainly warrants more attention to determine whether these sources can be quantified and remediated.

Giardia and *Cryptosporidium* in urban storm water are also a concern. There is evidence that urban watersheds and storm runoff might have higher concentrations of *Giardia* and *Cryptosporidium* than other surface waters (Stern, 1996). (See Table 8-10.) The primary sources of these pathogens are humans and wildlife. Although *Cryptosporidium* is found in less than 50 percent of storm water samples, data suggest that high *Cryptosporidium* values may be a concern for drinking water supplies. Both pathogens can cause serious gastrointestinal problems in humans (Bagley et al, 1998).

Table 8-10. Percentage Detection of *Giardia* Cysts and *Cryptosporidium* Oocysts in Subwatersheds and Wastewater Treatment Plant Effluent in the New York City Water Supply Watersheds

Source Water Sampled (No. of sources/ No. of samples)	Percent Detection			
	Total *Giardia*	Confirmed *Giardia*	Total *Cryptosporidium*	Confirmed *Cryptosporidium*
Wastewater effluent (8/147)	41.5	12.9	15.7	5.4
Urban subwatershed (5/78)	41.0	6.4	37.2	3.9
Agricultural subwatershed (5/56)	30.4	3.6	32.1	3.6
Undisturbed subwatershed (5/73)	26.0	0.0	9.6	1.4

Source: Stern et al, 1996.

Receiving Water Impacts

Fecal coliform bacteria, fecal streptococci, and *E. coli* are consistently found in urban storm water runoff. Their presence indicates that human or other animal waste is also present in the water and that other harmful bacteria, viruses, or protozoans might be present as well. Concentrations of these indicator organisms in urban storm water are highly variable even within a given monitoring site. Data for fecal coliform bacteria illustrate this variability: site concentrations range from 10 to 500,000 most probable number per 100 milliliters (MPN/100mL) (Schueler, 1999).

Concentrations in urban storm water typically far exceed the 200 MPN/100 mL threshold set for human contact recreation. The mean concentration of fecal coliform bacteria in urban storm water for 34 studies across the United States was 15,038 MPN/100mL (Schueler, 1999). Another national database of 1,600 samples (mostly Nationwide Urban Runoff Program data collected in the 1980s), estimates the mean concentration at 20,000 MPN/100 mL (Pitt, 1998). Fecal streptococci concentrations for 17 urban sites had a mean of 35,351 MPN/100 mL (Schueler, 1999). Transport occurs primarily as a result of direct surface runoff, failing septic systems, SSOs, CSOs, and illicit discharges.

Human health can be affected by bacterial impacts on receiving waters when bacteria standards for water contact recreation, shellfish consumption, or drinking water are violated. Epidemiological studies from Santa Monica Bay have documented frequent sickness in people who swim near outfalls (SMBRP, 1996). Documented illnesses include fever, ear infections, gastroenteritis, nausea, and flu-like symptoms. Table 8-11 describes the effects of bacteria and protozoan problems on different receiving waters.

Table 8-11. Effects of Bacteria on Receiving Waters

Resource Affected	Impacts and Citations
Streams	More than 80,000 miles of streams and rivers in non-attainment because of high fecal coliform levels (USEPA, 1998a)
Reservoirs	Increased treatment cost of drinking water due to bacteria contamination (USEPA, 1996)
Beaches	More than 4,000 beach closings or advisories (USEPA, 1998b)
Estuaries	Nearly 4% of all shellfish beds restricted or conditional harvest due to high bacteria levels (NOAA, 1992) and more than 4,000 beach closings or advisories (USEPA, 1998b)

8.2.3 PHYSICAL IMPACTS OF CONSTRUCTION AND LAND DEVELOPMENT ACTIVITIES

Construction and land development activities can have a number of impacts on stream systems, including impacts to stream hydrology, geomorphology, habitat structure, thermal regime, and direct channel impacts. These impacts are most visible on streams in urbanized areas. Construction and land development impacts on stream systems are described for each of these impact categories in Table 8-12. Because it is very difficult to differentiate between physical impacts that occur during construction and impacts that result from postdevelopment conditions, the discussion addresses physical impacts from a broader perspective. It does not differentiate between short-term effects arising and site construction activities from long-term impacts of postdevelopment conditions.

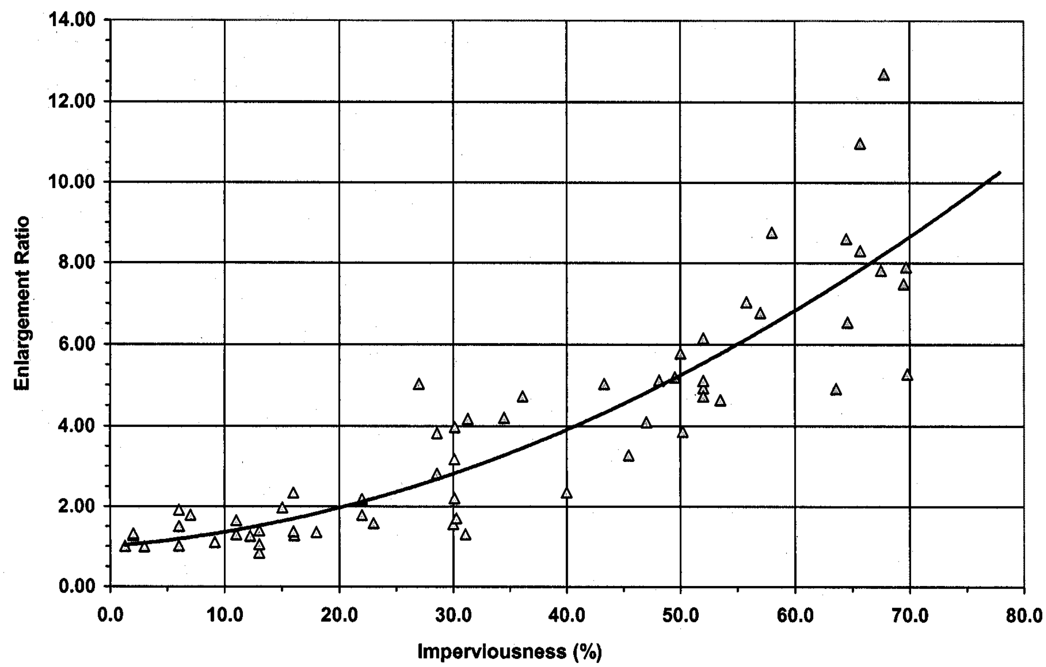
Physical changes are often precipitated by changes in hydrology that result when permeable rural and forest land is converted to impervious surfaces like pavement and rooftops and relatively impermeable urban soils. The conversion causes a fundamental change in the hydrologic cycle because a greater fraction of rainfall is converted to surface runoff. This change in the basic hydrologic cycle causes a series of other impacts (Table 8-12). The stream immediately begins to adjust its size, through channel erosion, to accommodate larger flows. Streams normally increase their cross-sectional area by incising, widening, or often both. This process of channel response to increases in impervious surfaces accelerates sediment transport and destroys habitat. In addition, urbanization frequently requires alteration of natural stream channels, such as straightening or lining with concrete or rock to transport water away from developed areas more quickly. Finally, impervious surfaces also absorb heat, thereby increasing stream temperatures during runoff events.

Table 8-12. Physical Impacts of Urbanization on Streams

Impact Class	Specific Impacts	Cause(s)
Hydrologic	• Increased runoff volume • Increased peak flood flow • Increased frequency of "bankfull" event • Decreased baseflow	• Paving over natural surfaces • Compaction of urban soils
Geomorphic	• Sediment transport modified • Channel area increase to accommodate larger flows	• Modified flows • Channel modification • Construction
Habitat structure	• Stream embeddedness • Loss of large woody debris • Changes in pool/riffle structure	• Modified flows • Stream channel erosion • Loss of riparian area
Thermal	• Increased summer temperatures	• Heated pavement • Storm water ponds • Loss of riparian area
Channel modification	• Channel hardening • Fish blockages • Loss of first and second order streams through storm drain enclosure	• Direct modifications to the stream system.

Figure 8-2 (Claytor and Brown, 2000; MacRae and De Andrea, 1999) depicts the impacts of land development on the stream channel. At low levels of imperviousness, the stream has a stable channel, contains large woody debris, and has a complex habitat structure. As urbanization increases, the stream becomes increasingly unstable, increases its cross-sectional area to accommodate increased flows, and loses habitat structure. In highly urbanized areas, stream channels are often modified through channelization or channel hardening. These physical changes are often accompanied by decreased water quality.

Figure 8-2. Stream Channel Enlargement as a Function of Watershed Imperviousness



8.2.3.1 Hydrologic Impacts of Construction and Land Development Activities

The increased runoff volume that results from land development alters the hydrograph from its predeveloped condition. The resulting hydrograph accommodates larger flows with higher peak-flow rates. Because storm drain conveyance systems (e.g., curbs, gutters) improve the efficiency with which water is delivered to the stream, the hydrograph is also characterized by a more rapid time of concentration and peak discharge. Finally, the flow in the stream between events can actually decrease because less rainfall percolates into the soil surface to feed the stream as baseflow. The resulting hydrologic impacts include increased runoff volume, increased flood peaks, increased frequency and magnitude of bankfull storms, and decreased baseflow volumes.

Increased Runoff Volume

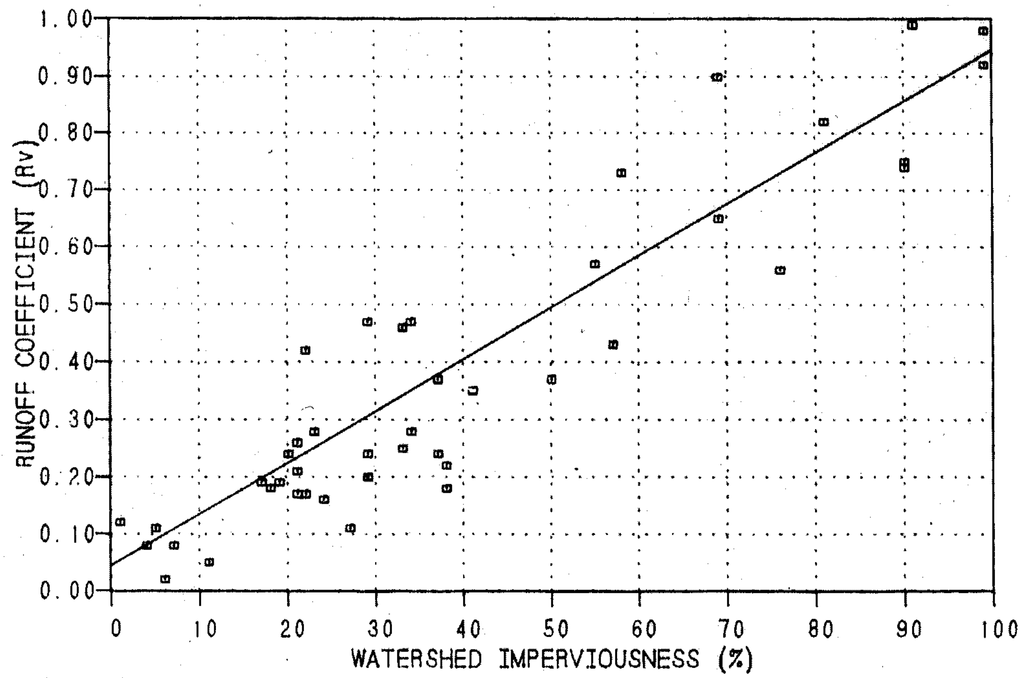
Impervious surfaces and urban land use changes alter infiltration rates and increase runoff volumes. Table 8-13 shows the difference in runoff volume between a meadow and a parking lot. The parking lot produces approximately 15 times more runoff than a meadow for the same storm event. Schueler (1987) demonstrated that runoff values increase significantly with the impervious surfaces in a watershed (Figure 8-3). The increased volume of water from urban areas is likely the greatest single cause of the negative impacts of urban storm water on receiving waters. The volume causes channel erosion and loss of habitat stability, as well as an increase in the total load of many pollutants such as sediment and nutrients.

Table 8-13. Hydrologic Differences Between a Parking Lot and a Meadow

Hydrologic or Water Quality Parameter	Parking Lot	Meadow
Runoff coefficient	0.95	0.06
Time of concentration (minutes)	4.8	14.4
Peak discharge, 2-yr, 24-h storm (ft^3/s)	4.3	0.4
Peak discharge rate, 100-yr storm (ft^3/s)	12.6	3.1
Runoff volume from 1-in. storm (ft^3)	3,450	218
Runoff velocity @ 2-yr storm (ft/sec)	8	1.8

Key Assumptions: 2-yr, 24-hr storm = 3.1 in.; 100-yr storm = 8.9 in.
Parking Lot: 100% imperviousness; 3% slope; 200-ft flow length; hydraulic radius = 0.03; concrete channel; suburban Washington ‘C’ values
Meadow: 1% impervious; 3% slope; 200-ft flow length; good vegetative condition; B soils; earthen channel
Source: Schueler, 1987.

Figure 8-3. Runoff Coefficient as a Function of Impervious Cover



NOTE: 44 small urban catchments monitored during the national NURP study.

Construction activities also cause fundamental modifications in native soils. The compaction of urban soils and the removal of topsoil during construction decreases the infiltration capacity of the soil, resulting in a corresponding increase in runoff (Schueler, 2000). The bulk density is a measure of soil compaction, and Table 8-14 shows the values for different aspects of urbanization.

Table 8-14. Comparison of Bulk Density for Undisturbed Soils and Common Urban Conditions

Undisturbed Soil Type or Urban Condition	Surface Bulk Density (grams/cubic centimeter)
Peat	0.2 to 0.3
Compost	1.0
Sandy Soils	1.1 to 1.3
Silty Sands	1.4
Silt	1.3 to 1.4
Silt Loams	1.2 to 1.5
Organic Silts/Clays	1.0 to 1.2
Glacial Till	1.6 to 2.0
Urban Lawns	1.5 to 1.9
Crushed Rock Parking Lot	1.5 to 1.9
Urban Fill Soils	1.8 to 2.0
Athletic Fields	1.8 to 2.0
Rights of Way and Building Pads (85%)	1.5 to 1.8
Rights of Way and Building Pads (95%)	1.6 to 2.1
Concrete Pavement	2.2

Note: Shading indicates "urban" conditions.
Source: Schueler, 2000.

Increased Flood Peaks

Increased surface runoff following urbanization increases peak flows. Data from Sauer et al (1983) suggest that peak flow from large flood events (10-year to 100-year storm events) increases substantially with urbanization. The paper presents results of a survey of urban watersheds throughout the United States and predicts flood peaks based on watershed impervious cover and a "basin development factor" that reflects watershed characteristics such as the amount of curb and gutter, and channel modification. These data suggest that at 50 percent impervious cover, the peak flow for the 100-year event can be as much as twice that in an equivalent rural watershed. Data from Seneca Creek in Montgomery County, Maryland, suggest a similar trend. The watershed experienced significant growth during the 1950s and 1960s. Comparison of gauge records from 1961 to 1990 to those from 1931 to 1960 suggests that the peak 10-year flow event increased from 7,300 to 16,000 cfs, an increase of more than 100 percent (Leopold, 1994).

Increased Frequency and Volume of Bankfull Flows

Stream channel morphology is more influenced by frequent (1- to 2-year) storm events, or "bankfull" flows, than by large flood events. Hollis (1975) demonstrated that urbanization increased the frequency and magnitude of these smaller-sized runoff events much more than the larger events. Data from this study suggest that streams increase their 2-year bankfull discharge by two to five times after development takes place. Many other studies have documented the increase in flow associated with impervious cover. A study by Guay (1995) compared the 2-year flow events before and after development in an urban watershed in Parris Valley, California, in the 1970s and in the 1990s. The impervious level of 9 percent in the 1970s increased to 22.5 percent by the 1990s. The 2-year discharge more than doubled from 646 cfs to 1,348 cfs. A 13 percent change in impervious cover resulted in a doubling of the 2-year peak flow.

A significant impact of land development is the frequency with which the bankfull event occurs. Leopold (1994) observed a dramatic increase in the frequency of the bankfull event in Watts Branch, an urban subwatershed in Rockville, Maryland. This watershed also experienced significant development between the 1950s and 1960s. A comparison of gauge records indicated that the bankfull storm event frequency increased from two to seven times per year from 1958 to 1987.

Changes in Baseflow

Land development results in a smaller recharge to groundwater and a corresponding decrease in stream flow during dry periods (baseflow). Only a small amount of evidence, however, documents this decrease in baseflow. Spinello and Simmons (1992) demonstrated that baseflow in two urban Long Island streams went dry seasonally as a result of urbanization. Another study in North Carolina could not conclusively determine that urbanization reduced baseflow in some streams in that area (Evett et al, 1994). It is important to note, however, that groundwater flow paths are often complex. Water supplying baseflow feeding the stream can be from deeper aquifers or can originate in areas outside the surface watershed boundary. In arid and semiarid areas, watershed managers have reported that baseflow actually increases in urban areas. Increased infiltration from people watering their lawns and return flow from sewage treatment plants are two possible sources (Caraco, 2000). Recharge of clean groundwater is important in these communities, and managers would rather see clean water infiltrated than transported as surface water during storm events.

8.2.3.2 Impacts on Geomorphology/Sediment Transport

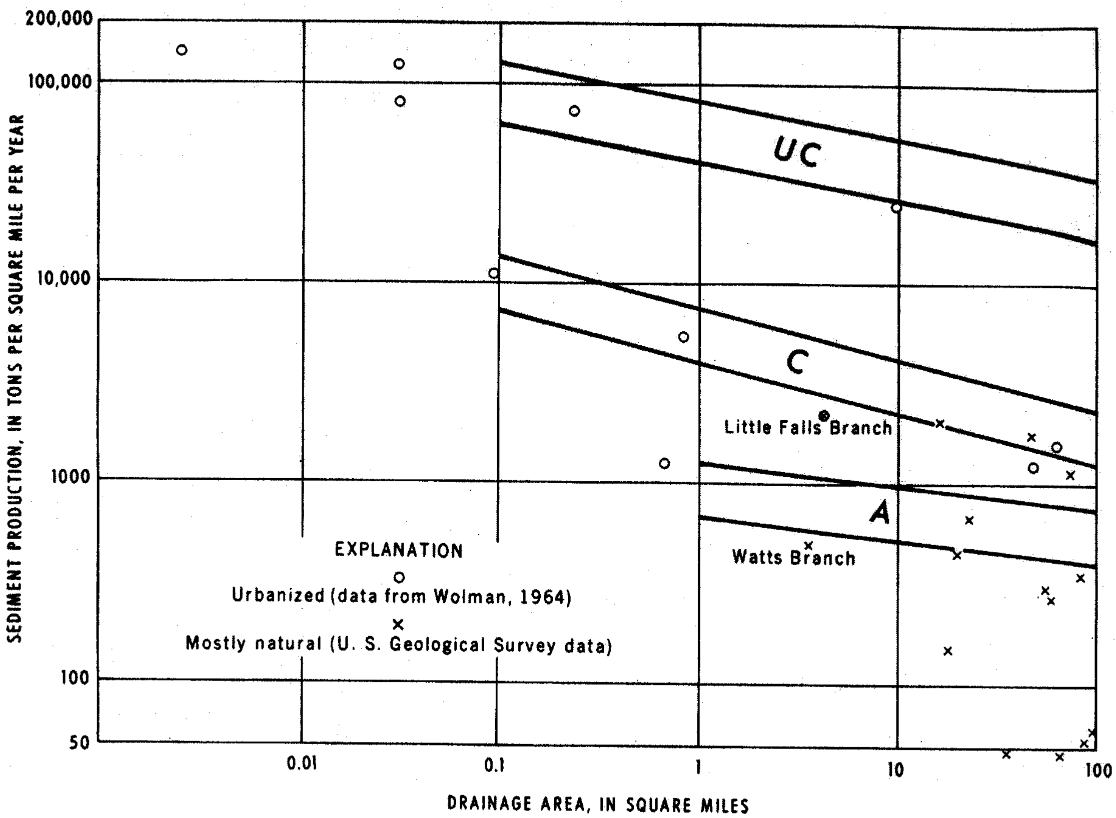
Changes in hydrology, combined with additional sediment sources from construction and modifications to the stream channel, result in changes to the geomorphology of stream systems. These impacts include increased, and sometimes decreased, sediment transport and channel enlargement to accommodate larger flows.

Increased Transport of Sediment

The increased frequency of bankfull (1- to 2-year) storms causes more "effective work" (as defined by Leopold), causing greater sediment transport and bank erosion to take place within the channel. For the same storm event, the increased volume results in a greater amount of total stress above the critical shear stress required to move bank sediment (Schueler, 1987). This effect is compounded by the fact that smaller, more frequent storm events also cause flows in excess of the stress required to move sediment.

The result of this change in effective work on streambanks is increased channel erosion. Studies in California (Trimble, 1997) and Austin, Texas (Dartinguenave et al, 1997) suggest that 60 to 75 percent of the sediment transport in urban watersheds is from channel erosion as compared to estimates of between 5 percent and 20 percent for rural streams (Collins et al, 1997; Walling and Woodward, 1995). If the sediment is not deposited in the channel at obstructions, it is transported downstream to receiving waters such as lakes, estuaries, or rivers. The result can be reduced storage and loss of habitat due to the filling of these water bodies. The clearing and grading of land for new construction at the outset of urbanization is another source of sediment in urban streams. Figure 8-4 (Leopold, 1968) illustrates the difference in sediment from uncontrolled and controlled construction sites.

Figure 8-4. Sediment Production from Construction Sites



Annual sediment production per square mile for urbanized and natural areas. Zones: *A*, agricultural; *C*, under construction; *UC*, under construction and undiluted.

Decreased Sediment Transport

Decreased sediment transport off the land surface itself can result after urbanization as natural drainage and first-order channels are replaced by storm drains and pipes (Dunne and Leopold, 1978). Channel erosion downstream might result when any export of sediment is not replaced by diminished upstream sediment supply. Ultimately, after significant erosion has taken place, the downstream channel will have adjusted to its postdevelopment flow regime and sediment transport will be reduced. Hence, the stability of the land surface and the piping of drainage channels limit storm water's exposure to sediment and reduce the sediment supply.

Increase in Size of Channel

Channels increase their cross-sectional area to respond to higher and more frequent urban flows. In postdevelopment urban watersheds, the increase in frequency of this channel-forming event normally causes sediment transport to be greater than sediment supply. The channel widens (and/or downcuts) in response to this change in sediment equilibrium (Allen and Narramore, 1985; Booth, 1990 Hammer, 1977; Morisawa and LaFlure, 1979;). Some research suggests that over time channels will reach an "ultimate enlargement," relative to a predeveloped condition, and that impervious cover can predict this enlargement ratio (MacRae and DeAndrea, 1999). This was shown in Figure 8-2, which depicted the relationship between ultimate stream channel enlargement and impervious cover for alluvial streams, based on data from Texas, Vermont, and Maryland.

Stream channels expand by incision, widening, or both. Incision occurs when the stream down-cuts and the channel expands in the vertical direction. Widening occurs when the sides of the channel erode and the channel expands horizontally. Either method results in increased transport of sediment downstream and degradation of habitat. Channel incision is often limited by grade control from bedrock, large substrate, bridges, or culverts. These structures impede the downward erosion of the stream channel and limit incision. In substrates such as sand, gravel, and clay, however, stream incision can be of greater concern (Booth, 1990).

Channel widening more frequently occurs when streams have grade control and the stream cuts into its banks to expand its cross-sectional area. Urban channels frequently have artificial grade control due to the frequent culverts and road crossings. These are often areas where sediment can accumulate as a result of undersized culverts and bridge crossings.

8.2.3.3 Changes in Habitat Structure

Land development results in many changes in habitat structure, including embeddedness, decreased riffle/pool quality, and loss of large woody debris (LWD). Increased sedimentation due to clearing and grading during construction as well as bank erosion can significantly reduce the amount of habitat for substrate-oriented species.

Increased sediment transport from construction and land development can fill the interstitial spaces between rocks and riffles, which are important habitat for macroinvertebrates and fish species, such as darters and sculpins. The stream bottom substratum is a critical habitat for trout and salmon egg incubation and embryo development (May et al, 1997).

The presence and stability of LWD is a fundamental habitat parameter. LWD can form dams and pools, trap sediment and detritus, provide stabilization to stream channels, dissipate flow energy, and promote habitat complexity (Booth et al, 1996). For example, depending on the size of the woody debris and the stream, the debris can create plunge, lateral, scour, and backwater pools, short riffles, undercut banks, side channels, and backwaters, and create different water depths (Spence et al, 1996). The runoff generated in urban watersheds from small storms can be enough to transport LWD. Maxted et al (1994) found that woody debris were typically buried under sand and silt in urban streams. In addition, the clearing of riparian vegetation limits an important source of large woody debris. Horner et al (1997) present evidence from the Pacific Northwest that illustrates LWD in urban streams decreases with increased imperviousness.

Habitat diversity is a key factor in maintaining a diverse and well-functioning aquatic community. The complexity of the habitat results in increased niches for aquatic species. Sediment and increases in flow can reduce the residual depths in pools and decrease the diversity of habitat features such as pools, riffles, and runs. Richey (1982) and Scott et al (1986) reported an increase in the prevalence of glides and a corresponding altered pool/riffle sequence due to urbanization.

8.2.3.4 Thermal Impacts

Summer in-stream temperatures have been shown to increase significantly (5 to 12 degrees) in urban streams because of direct solar radiation, runoff from heat-absorbing pavement, and discharges from storm water ponds (Galli, 1991). Increased water temperatures can prevent temperature-sensitive species from surviving in urban streams.

Water temperature in headwater streams is strongly influenced by local air temperatures. Galli (1991) reported that stream temperatures throughout the summer are higher in urban watersheds, and the degree of warming appears to be directly related to the imperviousness of the contributing watershed. Over a 6-month period, five headwater streams in the Maryland Piedmont that have different levels of impervious cover were monitored. Each urban stream had mean temperatures that were consistently warmer than that of a forested reference stream, and the size of the increase appeared to be a direct function of watershed imperviousness. Other factors, such as a lack of riparian cover and ponds, were also shown to amplify stream warming, but the primary contributing factor appeared to be watershed impervious cover.

8.2.3.5 Direct Channel Impacts

Channel Straightening and Hardening/Reduction in First-Order Streams

Channel straightening and hardening includes the addition of riprap or concrete to the channel, the straightening of natural channels, and the piping of first-order and ephemeral streams. Although this conversion process is often done to control runoff from urbanized areas, adverse impacts often occur downstream. In a national study of urban watersheds in 269 gauged basins, Sauer et al (1983) determined that channel straightening and channel lining (hardening)–along with the percentage of curbs and gutters, streets, and storm sewers–were the dominant land use variables affecting storm flow. These variables all affect the efficiency with which water is transported to the stream channel. Maintaining this efficiency increases the velocities needed for storm water to exceed critical shear stress velocities, eroding the channel. These factors also considerably degrade any natural habitat for stream biota.

Embedded Sediment

Sediment embeddedness measures the degree to which cobbles and large gravels are buried and their interstitial spaces filled because of fine sediment deposition. In a study of habitat restoration in a highly sedimented Idaho stream, Hillman et al (1987) found that interstitial spaces among cobbles may be essential winter habitat for juvenile chinook salmon. When large cobble was added to an otherwise embedded stream, juvenile populations increased. When that same cobble became embedded, the population decreased.

Embeddedness blocks passages and removes small cover spaces for eggs, fry and juvenile fish. USEPA (2003) summarized that sediment deposition has caused a 94% reduction in numbers and standing crop biomass in large game fish due to increased vulnerability of their eggs to predation in gravel and small rubble, reductions in oxygen supply to eggs, and increased embryo mortality.

Weaver and Fraley (1993) (in USEPA 2003) reported that emergence success of cutthroat trout was reduced from 76% to 4% when fine sediment was added to redds. NAHB (2000) reported that as fry grow into juvenile fish they seek out the slow moving water at the channel edges for cover. These areas also are favored for deposition of suspended sediment. When these areas are filled with excess sediment, sheltered space is lost and the juveniles are forced out into the channel to compete at a disadvantage with the adult fish. Waters (1995) also found that juveniles face habitat degradation from the sedimentation of the pools. Information quantitatively relating embeddedness levels to effects on aquatic fauna is limited.

NAHB (2000) found that invertebrate study results are often complicated by the fact that the various invertebrate species in a community responds very differently to increased sediments. Aquatic insect densities may decline at embeddedness levels of approximately two-thirds to three-quarters.

Surface Sediment

Surface sediment describes the percentage of streambed area with exposed fine sediments. Targets are developed to describe thresholds of suitability of stream substrates for invertebrate and salmonid habitation. Using the Wolman pebble count method, percent surface fines may be calculated. The same method is also used to determine the median substrates size (d50). This is used as a sediment target. The percentage of area is one measure, but particle size distribution, geometric mean particle size, median particle size, or other indices like fredle index may be used to describe the streambed's exposed fine sediment area.

Salmonids prefer mid-sized substrates with interstitial cover to either fine sediment or boulders and bedrock. Ephemeroptera, Plecoptera, and Trichoptera (important fish-food organisms) also respond positively to gravel and cobble substrates (Waters 1995). However, the percent coverage of fine sediments by area and the effects on salmonids and invertebrates have not been extensively investigated.

NAHB (2000) found a notable absence of data regarding effects of suspended sediments on warmwater fish. They also found evidence that some warmwater fish may be able to spawn on muddy substrate. Studies on the effects of surface sediment from construction activities are limited. However, one study by Reed (1977) in Wheeler et al (2003) did reveal that sediment from road construction in Northern Virginia reduced aquatic insect and fish communities by up to 85 percent and 40 percent, respectively.

Subsurface Sediment

Surface fines and embeddedness are apparent to the human observer, and are thus relatively easy to measure, but subsurface or depth fines also have a major effect on the suitability of spawning habitats. The amount of subsurface fine sediments as measured at the head of riffles in likely spawning areas can be an indication of redd site suitability, conditions for egg survival and alevin emergence in the constructed redd, as well as habitat quality for fry and prey.

Information on the biological effects of subsurface sediment varies according to the size of sediment and geographic area of concern. Some of the variability is reduced by standardizing the habitat and stream types (e.g., Rosgen [1996] level II) sampled. Subsurface sediment targets can serve as a measure of suitability for fish spawning grounds, and they are most applicable in riffles and spawning areas in streams with gravel/cobble/boulder streambeds. If there are excessive subsurface fines they can have detrimental effects on salmonid and invertebrate habitat suitability and redd conditions. In the western U.S. redd construction is often upstream from riffles or at the tail end of pools where there is a net flow of stream water downward into the substrate. Where upwelling groundwater rather than surface irrigates the substrate, the fines are no longer in the position to block the flow of water into the redd, and therefore are a less important threat (Waters, 1995).

Riffle Stability

The Riffle Stability Index (RSI) indicates the relative percentage of the streambed that is mobile during channel forming flows. Bed mobility is related to pool quality and abundance. With lower RSI values, there is overall greater residual pool volume, because less of the streambed is susceptible to moving. Pool habitat provides critical refuge for juvenile and adult salmonids. The RSI has been used as an indicator of beneficial use, especially as related to cold water biota. The RSI is measured as the percentage of the substrate particles (from a Wolman pebble count) that are smaller than the largest particles that are moved in channel forming flows. Particles on point bars are measured to determine the largest mobile particles.

Intergravel Dissolved Oxygen

One effect of the accumulation of fine sediment in the aquatic environment is reduced permeability of the substrate resulting in less oxygen exchange to support fish embryos and macroinvertebrates. Salmonids excavate streambed substrate to deposit eggs then backfill the “egg pocket” to protect the eggs during the incubation period. The eggs are dependent on the flow of oxygen-rich water through the substrate to survive. The accumulation of fines in the redd restricts water flow and reduces oxygen to the eggs which results in decreasing survival (Shapovalov and Berrian, 1939; Wickett, 1954; Shelton and Pollock, 1966).

Several studies have related intergravel dissolved oxygen to egg/fry survival. Survival of embryos has been positively correlated with intergravel dissolved oxygen in the redds for steelhead (Coble, 1961) and brown trout (Maret et al, 2003). Silver et al (1963) found that embryos incubated at low and intermediate DO concentrations produced smaller and weaker alevins than embryos incubated at higher concentrations. Weak sac fry cannot be expected to survive rigorous natural conditions. In a review of embryo development studies, Chapman (1988) noted several examples of developmental impairment at lower DO concentrations, but did not recommend a single threshold. Bjornn and Reiser (1991) recommended that intergravel DO concentrations should be at or near saturation, and that temporary reductions should drop to no lower that 5.0 mg/L.

Observations of the effects of intergravel flow on macroinvertebrates are much less extensive than those for fish. Excessive sediment affects macroinvertebrates by accumulating on the body surfaces and reducing the effective area of the respiratory structures (Lemly, 1982) or by covering pupae cases and reducing the flow of oxygenated water to the metamorphosing insect (Rutherford and Mackay, 1986).

Fish Blockages

Infrastructure associated with urbanization–such as bridges, dams, and culverts–can have a considerable effect on the ability of fish to move freely upstream and downstream in the watershed. This in turn can have localized effects on small streams, where nonmigratory fish species can be inhibited by the blockage from recolonizing areas after acutely toxic events. Anadromous fish species such as shad, herring, salmon, and steelhead also can be blocked from making the upstream passage that is critical for their reproduction.

8.2.3.6 Site Differences in Physical Impacts

Site differences that can affect physical impacts include location of the impervious surfaces, presence of vegetation, and soil type within the watershed. Location of the impervious development can be instrumental in the timing of runoff in a watershed. If the development is at the bottom of the watershed, peak flow from the urbanized area will likely have passed downstream before the flow peaks from the upper watersheds reach the urbanized area (Sauer et al, 1983). Vegetation can reduce channel erosion from storm flows. A study in British Columbia showed that meander bends with vegetation were five times less likely to experience significant erosion from a major flood than similar nonvegetated meander bends (Beeson and Doyle, 1995). The types and porosity of soils are also important in determining runoff characteristics from the land surface and erosion potential of the channels. Allen and Narramore (1985) showed that channel enlargement in chalk channels was from 12 to 67 percent greater than in shale channels near Dallas, Texas. They attributed the differences to greater velocities and shear stress in the chalk channels.

8.3 ANALYSIS OF SOIL TEXTURE BY REGION

EPA used surface soil texture as the primary indicator of soil nature for the 48 contiguous states. The USDA GIS coverage of surface soil texture (the top six inches of soil) was developed primarily to characterize agricultural areas. NRI (USDA, 2000) data indicates that agricultural land (crop land, pasture land and range land) makes up a large fraction of the land area converted to urbanized areas annually. The bulk of the remaining converted acreage is from areas characterized as forested. EPA used the agriculture-based USDA soil characterization data as a reasonable approximation of the soil texture that would be encountered on all new construction sites.

The USDA coverage also allowed for the identification of the three dominant soils for each ecoregion, listed in Table 8-15. Where more than three soils were present, only the top three textures were selected and the percentage of each prorated so that the total percentage equaled 100%. In each ecoregion the three dominant surface soil textures comprised at least 65 percent of the total surface area in each ecoregion when considering all soils present. This was judged to provide a reasonable approximation of the geographic distribution of construction site soils for each ecoregion. The per-ecoregion soil texture information was then subdivided into the state-ecoregion area basis for later use in computing erosion rates. In summary, the analysis identified seven different soil textures that dominate the surface soil coverage within the 48 contiguous states.

8.4 ESTIMATION OF SOIL EROSION RATES

The evaluation of soil erosion rates was based on previous procedures used by EPA to assess the environmental benefits of the Phase II Storm Water Rule (EPA, 1999), which utilized the Revised Universal Soil Loss Equation (RUSLE) (USDA, 1997). The pollutant of primary interest in storm water discharges from construction sites is sediment that results from eroded soil. This sediment is composed of both suspended solids (fine-grained material) and bedload (large-grained material). The analysis entailed evaluation of up to three dominant soils in each ecoregion (see Table 8-15), for three slopes (3, 7, and 12 percent). In this assessment, EPA assumed that construction sites were evenly divided among these three slopes. For all slope and soil combinations, the RUSLE equation was used to estimate the ambient annual erosion rate or yield (natural), and the erosion rate with construction activity occurring without any BMPs. These two erosion rates provide the basis for the estimate of loadings reductions related to implementation of construction site BMPs.

Table 8-15. Ecoregion Surface Soil Texture Characterization

Ecoregion	Soil #1 Texture	Percent Coverage	Soil #2 Texture	Percent Coverage	Soil #3 Texture	Percent Coverage
1	Sand	18.5%	Sandy Loam	34.2%	Loam	47.3%
2	Sand	11.3%	Sandy Loam	41.8%	Loam	46.9%
3	Sandy Loam	36.9%	Loam	63.1%		
4	Loamy Sand	29.0%	Sandy Loam	71.0%		
5	Sand	31.8%	Sandy Loam	51.2%	Loamy Sand	17.0%
6	Sand	78.2%	Loamy Sand	9.9%	Sandy Loam	11.8%
7	Sand	100.0%				
8	Sandy Loam	46.5%	Silt Loam	53.5%		
9	Silt Loam	62.6%	Sandy Loam	18.1%	Loam	19.4%
10	Silt Loam	54.0%	Sandy Loam	18.3%	Loam	27.7%
11	Silt Loam	59.7%	Sandy Loam	18.0%	Clay	22.3%
12	Silt Loam	54.0%	Sandy Loam	25.6%	Loam	20.4%
13	Silt Loam	31.5%	Loam	68.5%		
14	Sandy Loam	39.5%	Loam	60.5%		
15	Silt Loam	38.9%	Loam	61.1%		
16	Sandy Loam	52.4%	Loam	47.6%		
17	Silt Loam	37.5%	Loam	34.0%	Silty Clay	28.5%
18	Silt Loam	100.0%				
19	Sandy Loam	37.4%	Loam	43.2%	Loamy sand	19.4%

Within each of the 19 ecoregions, specific urban areas were selected as the areas where new construction is most likely to occur. Selecting specific urban areas was necessary in order to determine the appropriate rainfall characteristics and to set RUSLE equation parameters related to rainfall and soil cover. The erosion rates for these urban areas were assumed to be representative of the ecoregion as a whole. The specific urban areas analyzed within each ecoregion are presented in Table 8-16. This table also presents the range of sediment yields for the three slopes and dominant soils in each ecoregion. When computing the values in Table 8-16, the role of construction site BMPs were not considered–the estimates are solely ambient conditions and disturbed (denuded) conditions. BMP removal rates are discussed in Section 8.5.

As shown in Table 7-4, it was assumed that some portion of each construction site will remain undisturbed, depending on site size and ultimate land use. This is due to a certain percentage of each site comprising features such as open space, natural area set-asides, stream buffers, and forested buffers. For the estimated fraction of each construction site expected to be undisturbed, EPA set the rate of eroded material to ambient levels. For example, disturbed sand soils in ecoregion 1 produce a maximum construction site yield of 2.71 tons per acre, and undisturbed sand soils on construction sites will produce 0.69 tons per acre.

The duration of construction site activities and timing of these activities are variables that affect how much eroded soil is generated. Several factors are simplified in this assessment in order to avoid complexity and the use of excessive analytical resources. First, the assumed length of the construction period spans a calender year, regardless of construction site size, meaning there is no "wintering over" of partially constructed areas. Since the estimates of construction acreage are based on annual values obtained from NRI, this is a reasonable assumptions. Although large construction projects will likely span several years, the basis of the analysis is the amount of acreage actually being developed in any given year.

The timing of construction activities (e.g., clearing and grubbing) are assumed to occur in ways that minimize soil erosion. Instead of denuding an entire large site at a single time, construction operators are assumed to used a phased approach to land disturbance, where only portions of each construction site are cleared and graded before moving on to other portions. EPA acknowledges this assumption will likely result in underestimating the actual loadings, as it neglects the fact that large portions of the site may be disturbed for a considerable period of time.

Table 8-16. Range of Annual Erosion Estimates by Dominant Soil Type in Each Ecoregion (tons/acre/year)

Soil Type	Minimum Ambient Yield	Maximum Ambient Yield	Minimum Construction Site Yield	Maximum Construction Site Yield
Ecoregion 1, Indicator City Denver, Co				
Sand	0.19	0.69	0.73	2.71
Sandy Loam	1.01	3.73	3.96	14.63
Loam	1.42	5.25	5.58	20.59
Ecoregion 2, Indicator City Salt Lake, Ut				
Sand	0.07	0.26	0.36	1.33
Sandy Loam	0.38	1.39	1.95	7.20
Loam	0.53	1.96	2.74	10.13
Ecoregion 3, Indicator City Austin, Tx				
Sandy Loam	12.13	44.76	29.46	108.73
Loam	17.07	63.00	41.46	153.03
Ecoregion 4, Indicator City Atlanta, Ga				
Loamy Sand	5.26	19.41	13.87	51.20
Sandy Loam	11.83	43.67	31.21	115.20
Ecoregion 5, Indicator City Charleston, SC				
Sand	3.13	11.57	8.00	29.54
Sandy Loam	16.92	62.46	43.22	159.51
Loamy Sand	7.52	27.76	19.21	70.89
Ecoregion 6, Indicator City Jacksonville, Fl				
Sand	3.92	14.46	10.00	36.92
Loamy Sand	9.40	34.70	24.01	88.62
Sandy Loam	21.15	78.08	54.02	199.39
Ecoregion 7, Indicator City Miami, Fl				
Sand	5.22	19.28	13.34	49.23
Ecoregion 8, Indicator City Albany, NY				
Sandy Loam	3.33	12.30	10.35	38.21
Silt Loam	5.93	21.87	18.40	67.92
Ecoregion 9, Indicator City Pittsburgh, Pa				
Silt Loam	9.18	33.90	28.53	105.28
Sandy Loam	5.17	19.07	16.05	59.22
Loam	7.27	26.83	22.58	83.35

Soil Type	Minimum Ambient Yield	Maximum Ambient Yield	Minimum Construction Site Yield	Maximum Construction Site Yield
Ecoregion 10, Indicator City St. Paul/Minneapolis				
Silt Loam	5.4	20.01	21.03	77.62
Sandy Loam	3.05	11.25	11.83	43.66
Loam	4.29	15.84	16.65	61.45
Ecoregion 11, Indicator City Houston, Tx				
Silt Loam	35.94	132.63	87.29	322.17
Sandy Loam	20.21	74.61	49.10	181.22
Clay	9.73	35.92	23.64	87.25
Ecoregion 12, Indicator City Kansas City, Mo				
Silt Loam	12.61	46.55	36.15	133.40
Sandy Loam	7.09	26.18	20.33	75.04
Loam	9.98	36.85	28.62	105.61
Ecoregion 13, Indicator City Rapid City, SD				
Silt Loam	2.02	7.46	7.93	29.26
Loam	1.60	5.91	6.28	23.16
Ecoregion 14, Indicator City Boise, Id				
Sandy Loam	0.20	0.75	1.16	4.27
Loam	0.29	1.05	1.63	6.01
Ecoregion 15, Indicator City Eureka, Ca				
Silt Loam	4.55	16.78	17.20	63.49
Loam	3.60	13.28	13.62	50.27
Ecoregion 16, Indicator City San Francisco, Ca				
Sandy Loam	1.21	4.47	4.58	16.92
Loam	1.70	6.29	6.45	23.81
Ecoregion 17, Indicator City: Olympia/Seattle, Wa				
Silt Loam	3.35	12.36	12.68	46.78
Loam	2.65	9.79	10.04	37.04
Silty Clay Loam	2.58	9.53	9.77	36.06
Ecoregion 18, Indicator City: Spokane/Highland, Wa				
Silt Loam	0.30	1.11	1.71	6.32

Soil Type	Minimum Ambient Yield	Maximum Ambient Yield	Minimum Construction Site Yield	Maximum Construction Site Yield
Ecoregion 19, Indicator City: Stampede Pass/Mount Hood, Wa				
Sandy Loam	1.35	4.97	5.09	18.80
Loam	1.89	6.99	7.17	26.46
Loamy sand	0.60	2.21	2.26	8.35

Another assumption made in the analysis is that the size distribution of eroded material matches the native (dominant) soils. Table 8-17 indicates the grain size distribution of seven common soil textures believed to be present at a majority of construction sites.

Table 8-17. Estimated Soil Grain Size Distribution

Gross Soil Texture Classification	Clay Fraction, %	Fine Silt Fraction, %	Silt Fraction, %	Fine Sand Fraction, %	Sand Fraction, %
Clay	45	20	10	10	15
Loam	15	15	20	20	30
Loamy Sand	5.25	4.25	5	23.25	62.25
Sand	3.75	2.5	2.5	23.75	67.5
Sandy Loam	7.5	7.5	10	22.5	52.5
Silt Loam	11.25	18.75	30	21.25	18.75
Silty Clay Loam	18.75	22.5	32.5	18.75	7.5

Adapted from Foth, 1978

8.5 ESTIMATION OF BMP REMOVAL EFFICIENCIES

8.5.1 APPLICATION OF SEDCAD

BMP performance is dependent on many factors related to soil nature, hydrology, and engineering practice (see Section 5). A commercially available software package (SEDCAD) was used to model BMP removal efficiencies for a series of site conditions. These reference values were then used to estimate performance for each combination of soil, slope, location, and model construction site size, and reflecting the influence of the regulatory options considered on sediment discharges. Surface soil texture was the key feature used to adjust for the varying effects of soil nature on BMP removal efficiency.

BMPs were selected and sized for a subset of the model sites developed in Sections 4 and 7 reflecting the area draining to each BMP through the appropriate drainage pathway and following industry standard design practices. Table 8-18 provides an overview of the analysis performed, and detailed documentation of the specific design criteria and assumptions made can be found in the public record.

Table 8-18. Description of EPA Construction Site Analysis for BMP Removal Estimation

Item	BMP	Analysis Performed	Comments
Erosion Control	Seed and Mulch	No SEDCAD simulation	Estimation of removals was conducted in two phases. The first phase assumed that soils were exposed and unmanaged for varying periods of time to account for the active construction phase. In the second phase, soils were assumed to be stabilized with seed and mulch (see Table F-4). The total duration of each project was assumed to be 1 year.
Sediment Controls	Silt Fence	SEDCAD analysis generic to all model sites	
	Rock Check Dam	SEDCAD analysis generic to all model site sizes	
	Inlet Protection	SEDCAD analysis generic to all model site sizes	
	Sediment Trap	SEDCAD analysis of 3 acres of centralized drainage on a 7.5 acre model site	
	Sediment Basin	SEDCAD analysis of 10 acres of centralized drainage on a 25 acre model site	

For each BMP, performance was evaluated individually for 10 soil grain size groups under five different rainfall events ranging from 0.5 to 3.6 inches in depth. All NRCS or Soil Conservation Service (SCS) Type rainfall distributions (Type I, II, and III) were individually evaluated so that BMP performance would be customized to the climate on an ecoregion basis. For example, the estimated BMP performance within the relatively dry Ecoregion 1 is based on a range of rainfall events that are shaped according to the NRCS Type II distribution, or the rainfall distribution expected in the region.

The wide range in grain size groups was intended to improve the representativeness of the SEDCAD simulation of BMP performance to all of the likely conditions present across the country. This acknowledges that construction site BMPs have higher removal efficiencies for larger grained particles (such as sand) than for smaller grained particles (clays). By analyzing soil grain size groups individually, a reasonable basis for compositing an estimated removal rate was established for any of the common surface soil textures discussed in Section 8.3. BMPs are assumed to provide consistent performance for all sites that employ them in a single state-ecoregion area. For example, two sedimentation basins employed on a single 25 acre construction site were assumed to provide the same performance as four sedimentation basins employed on a 70 acre construction site. This simplification was necessary in order to limit the total analytical time and resources required to conduct the analysis.

8.5.2 CUSTOMIZING BMP REMOVALS FOR STATE-ECOREGIONS

The results of the basic SEDCAD analysis were used to develop BMP removal rates customized to each state-ecoregion area that reflect the role of:

- Dominant soils;
- Climate;
- Regulatory conditions (e.g,. baseline state regulations); and
- BMP combinations.

The suite of potential regulatory requirements includes stabilization of exposed soil areas within 14 days following the end of land disturbance. Because seeding (e.g., hydroseeding) and seeding with mulching for soil stabilization are common practices within the industry, it was assumed that this requirement would not increase the application rates of stabilization measures, but would rather only change the timing of stabilization. This was judged to be a reasonable assumption because existing state requirements include stabilization of exposed soils (although the time allotted may be 28 or 30 days instead or 14 days) or developers elect to stabilize exposed soils to prevent the need for subsequent re-grading.

EPA acknowledges there are difficulties involved in analyzing the effect of the shortened time period allowed for stabilization. Inherent to any analysis is uncertainty associated with the timing of land disturbing activities on various portions of a construction site, and further uncertainty related to seasonal variation in rainfall conditions across the country. So when developing its standardized approach within limits of its resources, EPA elected to focus on site physical features for a suite of model sites (e.g., site size, local soils texture) and the “typical” performance of seed/mulch as reported in the literature (derived from a range of soils and rainfall events).

To calculate the effects of seeding and mulching, EPA’s model assumed that well applied mulch provides the same sediment control effectiveness as grass. So, denuded construction surfaces are immediately stabilized as soon as the seed/mulch combination is applied. The idea behind this assumption is that as the mulch degrades, the grass germinates and grows, which then compensates for the loss of mulch. To estimate the sediments generated and released to the environment, the first step was to assign to each model site size category the period in the construction year that the site has bare soils or is covered by either mulch or grass (both for the baseline and regulatory conditions) (See Appendix F). For larger site sizes (larger than 1 acre), eroded site sediment generally goes through additional sediment control devices (e.g., sedimentation basins), whether site soils are bare or are stabilized with seed/mulch. So, for part of the construction year, seeding/mulching provides additional in-series control with downstream sediment controls within EPA’s suite of site models. The overall capture of eroded material for a construction year was set equal to the sum of the sediment captured in sediment control for the period without seed/mulch, plus the sediment captured by combined seed/mulch/sediment controls for the period following seed/mulch application. In this calculation, EPA's suite of site models estimates the generation of sediments on a per particle size basis (e.g., clay, silt, sand), based on local rainfall information and local (common) soils derived from national databases.

EPA acknowledges that this approach to estimating the influence of seeding and mulching on sediment discharges from construction sites likely underestimates the actual sediment discharges that will occur over the life of construction projects. Similarly, it also likely underestimates the reductions that will result from implementing soil stabilization within 14 days instead of 28 days since it ignores a number of important real-world factors. In addition, on residential projects individual lots are often sold off to a number of builders, and exposed soil areas are likely to persist for long periods of time in these areas. In addition, the analysis uses average or typical rainfall conditions. It ignores the influence of short-duration, high intensity storm events that could potentially occur throughout the construction project. However, despite its drawbacks, the analysis is reasonable given the analytical resources available in this case.

As shown in Table 8-17, each of the seven dominant surface soil textures can be characterized by the percent found in various grain size groups. The 10 soil grain size groups analyzed individually with SEDCAD provide key data for creating composited BMP removal rates for each dominant soil texture. Computing the amount of soil removed for a particular BMP is done by combining size-specific removals in proportion to the grain-size distribution of each soil. Step 1 in Table 8-19 presents the scale and purpose of the assessment of dominant soil grain size distributions.

The method for estimating construction site BMP removal rates in this analysis is probability-based, where the rainfall probability (i.e., the total rainfall depth occurring during an event) in each ecoregion is used to composite a probable annual performance for the model construction sites. Single-event BMP removal rates from SEDCAD were combined for each ecoregion to compute an "expected annual" removal rate. SEDCAD simulation of six individual rainfall events ranging from 0.5 to 5 inches in rainfall demonstrate how individual BMPs perform for various storm events. For each ecoregion, EPA analyzed ten years of precipitation records to categorize local rainfall patterns and estimate the probability that a storm of a given size will occur within a 1-year period (the assumed duration of construction projects). The expected annual removal value was then calculated for each BMP within each ecoregion from the cross-product of the BMP removal rate array with the ecoregion distribution of rainfall.

The expected value approach accounts for the fact that large but relatively infrequent events will have low removal rates (due to flows exceeding BMP design capacities and leading to bypasses, shorter detention times, or overtopping), while more frequent but smaller rainfall events will have higher removal rates. Step 2 in Table 8-19 indicates the scale and purpose of the probability-based assessment of ecoregion hydrologic characteristics. Table 8-20 indicates the range in soil-specific BMP removal rates for eroded construction site soils in the nineteen ecoregions.

The assessment of current state regulations (see Table 3-1) provides the basis for characterizing which of the seven model construction site sizes will employ a particular mix of BMPs under baseline conditions. The best example of this is a sediment trap for sites with between 5 and 10 acres of drainage area. Many states do not have this requirement as part of their existing program, but it is a requirement under Options 2 and 4. When calculating removal rates under baseline conditions for states without this requirement, removal rates were calculated for the 7.5 acre site group using BMP removal rates for rock check dams. For the analysis of BMP removals under Option 2 and 4, the removal rates were calculated using the more effective sediment trap.

Options 2 and 4 also affect construction site BMPs by way of setting minimum design requirements. The design basis for sediment basins under these options would increase from 1,800 to 3,600 cubic feet per acre of drainage for the 25 acre site size group in states that do not have this requirement under baseline conditions. The change in basin sizing would be reflected in the associated removal rate for those sites. Although many states do not specifically indicate minimum sediment basin requirements, EPA assumed that all construction sites of greater than 10 acres would implement sediment basins with at least 1,800 cubic feet per acre of storage, as basins are common practice in the industry. Step 3 in Table 8-19 indicates the scale and purpose of these considerations of current state regulations.

The combined performance of BMPs in series was assessed individually for each grain size group. An assumption was made that total BMP removal was equal to the removal from an erosion control BMP (i.e., seed and mulch), followed by the removal from a sediment control BMP (e.g, sediment trap). So, for 7.5 acre construction sites under Options 2 and 4, the total removal of clay-sized particles would be equal to the load of eroded clay-sized particles from the site, less the reduction of seed and mulch, and then less the estimated reduction of clay-sized particles provided by a silt trap. Step 4 in Table 8-19 indicates the scale and purpose of the BMP groupings used in this analysis.

Table 8-19. Methodology for Estimating BMP Removal Rates

SEDCAD Analysis	Other EPA Analysis	Examples
Step 1 - Soils Processing		
10 grain size groups covering from large sand to clay are individually analyzed, then combined to estimate individual removals for 3 major size groups; sand, silt, and clay	7 soil textures containing different amounts of sand, silt and clay were found to be common in the nation. SEDCAD output is used to estimate soil texture-specific removals, based on sand, silt, and clay fractions.	Loam texture soil contains 40, 40, and 20 percent sand, silt, and clay particles, respectively. SEDCAD lumped removals for these grain sizes in a sediment basin are 90, 40 and 10 percent, respectively, for a single rainfall event. The composited removal rate for the silt texture soils is calculated as 54 percent for the event.

SEDCAD Analysis	Other EPA Analysis	Examples
Step 2 - Precipitation Processing		
No direct role	For each size fraction, BMP removals are estimated for 6 rainfall events of increasing depth (0.5, 0.7, 1.2, 2.4, 3.6, and 5.0 inches of total precipitation) and then composited into a single expected removal rate.*	For the NRCS Type II rainfall distribution, SEDCAD sediment basin removal rates for the silt size fraction range from 100 (a 0.7-inch event) to 21 percent (a 5-inch event). The probability of the six rainfall events for Ecoregion 1 are used to composite an expected annual silt fraction removal rate of 98 percent for sediment basins. (Note, most rainfall events in the semi-arid Ecoregion I are small and fully retained within the wet storage portion of sediment basins with 3,600 cubic feet per acre of storage)
Step 3 - State Regulation Processing		
No direct role	For each area defined by the intersection of state and ecoregion boundaries, a decision is made on the presence or absence of BMPs under each option evaluated	A state found in Ecoregion 1 does not have a sediment basin requirement under baseline conditions. In this case, baseline reductions are based on removal rates of sediment basins with 1,800 cubic feet per acre of storage. Under Option 4, all sites would be required to install sediment basins with 3,600 cf/ac for large sites, so removal rates will range from 39% to 94% depending on the soils present.
Step 4 - BMP Combination		
No direct role	For centralized drainage and perimeter drainage (each), one erosion prevention BMP (e.g., seed/mulch) is followed by a single sediment control BMP (e.g,. sediment basin). The combined efficiency of the two BMPs is calculated individually for each land use and site size combination, which indicates the total removal.	For a loam soil, seed/mulch is 95 percent effective on all grain sizes. The remaining 5 percent enters a sediment basin where sand, silt, and clay size particles are individually assessed to determine the additional removal of each fraction. As a result, the combined removal of seed/mulch and sediment basins in a state in Ecoregion 1 is 98.5 percent (accounts for the probability of various rainfall events and the full capture with no discharge condition that occurs for frequent small events)

* Expected performance was based on all rainfall events encountered in 10 years of records for indicator cities selected for each ecoregion

Table 8-20. Range of BMP Percent Removals (Weighted by Grain Size Distribution)

Soil Texture	Silt Fence	Inlet Protection	Rock Check Dam	Silt Trap	Sediment Basin
Clay	34.5 / 40.4	17.3 / 20.8	17.3 / 20.8	30.9 / 39.8	38.8 / 62.2
Loam	67.4 / 73.3	34.2 / 39.4	34.2 / 39.4	59.9 / 70.8	64.2 / 79.7
Loamy Sand	89.9 / 91.5	67.1 / 72.2	67.1 / 72.2	88.0 / 90.9	89.3 / 93.8
Sand	93.5 / 94.3	72.4 / 77.5	72.4 / 77.5	92.5 / 94.1	93.4 / 96.1
Sandy Loam	83.7 / 86.6	57.2 / 62.4	57.2 / 62.4	80.0 / 85.4	82.1 / 89.8
Silt Loam	65.8 / 73.9	23.3 / 28.9	23.3 / 28.9	54.6 / 69.9	59.2 / 78.1
Silty Clay Loam	54.4 / 63.3	11.5 / 16.9	11.5 / 16.9	42.3 / 59.4	48.2 / 71.1
Range shows values across nineteen ecoregions					

8.6 CALCULATION OF NATIONAL LOADINGS AND REMOVALS BY REGULATORY OPTION

This assessment of model construction sites is intended to acknowledge major influences on national loadings, including site size, current state BMP requirements, soil nature, slopes and flow lengths of construction sites, and climate. Ultimately, the assessment resulted in 276 individual loadings estimates, which were combined with 9,000 individual estimates of BMP removal rates for various settings. For each state-ecoregion area, the analysis:

- Generated "whole site" estimates of the population of construction sites reflecting up to three dominant soils and three slopes (i.e, at no time were fractions of model construction sites analyzed);
- Estimated the amount of eroded soil produced due to construction activities on the basis of site size and land use type;
- Estimated BMP removal rates for the regulatory options;

Using the population of construction sites by land use and size (see Section 4.2.2), state-ecoregion area load totals based on the estimated load discharged from each model site were computed for baseline conditions and for each regulatory option. State-ecoregion area load totals were then summed to produce state and national total loads for each regulatory option (see Table 8-21). Tables F-1, F-2 and F-3 in Appendix F provide detailed information on loadings, including loadings to individual HUCs. Table 8-22 indicates estimated per-state loadings for each alternative. Note that the state-level loads in Table 8-22 do not sum to the national loads in Table 8-21 or 8-1 due to rounding.

Table 8-21. National Annual Construction Load Estimates

Site Size, acres	Single Family (tons)	Multi-family (tons)	Commercial (tons)	Industrial (tons)	Total (tons)
Baseline					
0.5	96,735	31,105	888,510	38,268	1,054,618
3	89,368	66,771	929,412	46,473	1,131,964
7.5	129,814	97,290	540,091	18,237	785,432
25	392,563	268,202	1,319,539	34,323	2,014,627
70	220,234	112,214	859,721	30,127	122,296
200	291,595	2,742	0	0	294,337
Total Load (tons)	1,220,308	578,325	4,537,274	167,428	6,503,334
Options 2 and 4					
0.5	96,735	31,105	888,510	38,268	1,054,618
3	89,368	66,771	929,412	46,473	1,132,024
7.5	73,781	55,356	306,504	10,608	446,249
25	315,661	215,635	1,061,546	28,068	1,620,910
70	183,283	94,895	716,892	26,102	1,021,172
200	246,314	2,151	0	0	248,465
Total Load (tons)	1,005,142	465,913	3,902,864	149,519	5,523,438
Option 2/4 Incremental Loading Reduction Estimate (tons)	215,166	112,412	634,410	17,909	979,896

Table 8-22. State Annual Construction Load Estimates (Tons)

State	Baseline	Options 2 and 4
AL	287,073	209,759
AR	170,647	170,647
AZ	31,901	31,901
CA	137,654	101,464
CO	10,713	6,882
CT	26,680	17,834
DE	13,992	12,789
FL	165,065	165,065
GA	402,299	346,641
IA	55,537	55,537
ID	4,988	4,988
IL	120,331	103,157
IN	109,407	93,942
KS	91,805	67,129
KY	164,311	152,279
LA	276,932	216,048
MA	58,414	58,414
MD	116,981	79,118
ME	34,821	34,821
MI	233,685	170,917
MN	157,401	115,099
MO	277,848	188,674
MS	295,241	216,221
MT	17,343	11,416
NC	358,486	263,116
ND	11,326	7,709
NE	38,323	28,211
NH	25,857	25,857
NJ	131,874	90,000
NM	32,418	32,418
NV	747	747
NY	118,749	118,749
OH	212,799	181,410
OK	134,039	134,039
OR	37,690	25,820
PA	346,182	273,585
RI	9,311	6,337
SC	146,239	146,239
SD	14,761	13,729
TN	323,505	323,505
TX	787,982	787,982
UT	4,258	4,258
VA	159,707	159,707
VT	8,113	5,561
WA	69,782	47,035
WI	158,684	106,885
WV	99,719	99,719
WY	3,251	2,338

8.7 INTEGRATION OF NATIONAL LOADINGS INTO NWPCAM

As described in Section 8.5, the analysis generated loadings for 146 state-ecoregion areas. State-ecoregion areas were created by overlaying state boundaries with the boundaries of the 19 EPA ecroegions. In order to determine HUC-level loadings, GIS processing was used to convert state-ecoregion loadings into loadings for the approximately 2,000 HUCs that span the 48 contiguous states. Individual HUCs were apportioned loads by overlaying state-ecoregion areas based on the development rate in the HUC obtained from NRI. For example, when two HUCs collectively cover a single state-ecoregion area the HUC with the highest rate of development is assumed to have a proportionately greater fraction of the state-ecoregion loadings than the neighboring lower-rate HUC.

Estimates of the number of construction sites within each HUC were based on the acreage developed within each HUC along with the distribution of construction sites by site size in Table 4-10. Numbers were rounded to whole numbers in order to prevent analytical problems associated with analyzing fractional sites. The per-site load within each HUC was calculated by dividing the total load for a site size group (i.e., 25 acres) by the number of sites in that site size category. The per-HUC construction site population and loadings were converted from GIS into a spreadsheet for subsequent analysis of benefits in NWPCAM. The HUC-level number of sites and associated loads are contained in Table F-3 of Appendix F. Note that due to rounding, the total number of sites and loads presented in Table F-3 do not match the national totals in Table F-1.

8.8 NWPCAM ASSESSMENT OF IN-STREAM SEDIMENT CONCENTRATIONS

8.8.1 NWPCAM SYSTEM OVERVIEW

The National Water Pollution Control Assessment Model (NWPCAM) is a national surface water quality model that simulates water quality improvements and economic benefits that result from water pollution control policies. NWPCAM is designed to characterize water quality for the nation’s network of rivers, streams, and lakes. NWPCAM incorporates a water quality model into a system designed for conducting national policy simulations and benefits assessments. NWPCAM is able to translate spatially varying water quality changes into willingness-to-pay values that reflect the value that individuals place on water quality improvements. In this way, NWPCAM is capable of deriving economic benefits estimates for a wide variety of water pollution control policies.

NWPCAM’s water quality modeling system is suitable for developing water quality estimates for virtually the entire inland portion of the country. Its national-scale framework allows hydraulic transport, routing, and connectivity of surface waters to be simulated in the 48 conterminous states. The model can be used to characterize source loadings (e.g., point sources) under a number of alternative policy scenarios (e.g., loadings with controls). These loadings are processed through the NWPCAM water quality modeling system to estimate in-stream pollutant concentrations on a detailed spatial scale and to provide estimates of policy-induced changes in water quality. The model incorporates routines to translate estimated concentrations into a six-parameter water quality index (WQI6) that provides a composite measure of overall water quality. The WQI6 allows for the calculation of economic benefits associated with the estimated water quality improvements. NWPCAM can be used to assess both the water quality impacts and the social welfare implications of alternative policy scenarios.

NWPCAM is an evolving system developed for EPA's Office of Water (OW) by RTI and has been used in several applications to estimate the benefits of pollution control policies. An adaptation of version 1.0 was used by OW's Office of Waste Management (OWM) to evaluate the potential benefits of the Stormwater Phase II rulemaking (Bondelid et al, 1999). Version 1.1 (RTI, 2000b), developed in response to external peer review on version 1.0, was oriented toward evaluating the effects of point source controls. NWPCAM version 1.1 was used in the proposed Meat Processing Effluent Guidelines rulemaking (EPA, 2003a). Version 1.5 was used in the proposed Animal Feed Operation/Confined Animal Feed Operation (AFO/CAFO) rulemaking (RTI, 2000a). Version 1.6 was used in developing the final AFO/CAFO rulemaking process (RTI, 2002). Version 2.1 with the Eutro-WASP kinetics model was used for analysis of the options for the construction and development final action. Complete documentation on NWPCAM and the modeling process used in this analysis can be found in RTI, 2004.

8.8.2 CONSTRUCTION AND LAND DEVELOPMENT MODELING PROCESS

8.8.2.1 Construction and Development Loads

The loads developed (see Tables 8-21, 8-22 and Tables F-1 and F-2 of Appendix F) for the options evaluated were distributed to the 8-digit hydrologic unit (HUC) level and broken out by site size. All loads were assumed to be TSS. These HUC-level loads are presented in Table F-3 in Appendix F. Loadings were developed for 1,717 HUCs for baseline conditions and the four regulatory options considered. Of the 1,717 HUCs, 57 (3%) were immediately excluded from the modeling analysis because they did not have an associated stream network in NWPCAM.

8.8.2.2 Distribution of Construction Sites and Loads

The methodology developed for distributing loads called for:

(1) Randomly distributing construction sites onto agricultural and forest land cover cells;

(2) Assigning loads to land cover cells based on the number and size of sites assigned to each land cover cell; and

(3) Removing background NPS TSS loads from land cover cells that were assigned construction sites based on the fraction of the cell that was covered by sites.

A total of 6,894,140 land cover cells were in the NWPCAM 2.1 database. Each land cover cell was assigned to one of eight general categories: agriculture, agriculture/herbaceous, agriculture/woodland, herbaceous, forest, water bodies/barren, tundra, and urban. Of the total land cover cells, 6,557,224 (95%) were assigned one of the first five land cover categories, and were classified as forest or agriculture. All of the forest and agricultural cells used during the site distribution process. Each agricultural and forest cell was assigned a random number.

An analysis was conducted to compare the construction site area against the available forest and agricultural land within each HUC. Of the 1,660 HUCs that had an associated stream network, 1,638 (95%) had at least as many agriculture/forested land cover cells as number of sites, indicating that no land cover cell would be assigned multiple construction sites. Six (<1%) had fewer agriculture/forest land cover cells than sites but had enough area to accommodate all sites. This indicated that some land cover cells were assigned multiple sites. Sixteen HUCs (1%) were excluded from the modeling analysis because they lacked land cover data.

Of the original 1,717 HUCs supplied in the loadings file, 1,644 HUCs were included in the final modeling analysis. Table 8-23 shows the total TSS loadings by loading option (i.e. mode run) for the 1,644 HUCs included in the modeling analysis. Of the 979,896 tons/year of loadings reductions estimated for Option 2/4, 941,108 tons/year, or 96%, were incorporated into the NWPCAM modeling.

Table 8-23. Summary of Construction and Development Loadings

Option	TSS Loading (ton/yr)
Baseline	6,288,751
Options 2 and 4	5,347,643

A computer module was used to distribute construction sites onto land cover cells. For each HUC, the module selected its associated land cover cells, ordered by the random identification numbers. Sites were distributed by assigning each land cover cell one site before moving on to the next land cover cell. When there were more sites than land cover cells, the code went back to the first land cover cell on the list and continued looping until all sites were distributed. The sites were distributed in order of decreasing size: 200 acres, 70 acres, 25 acres, 7.5 acres, 3 acres, and 0.5 acres. Since the land cover cells were randomly ordered, this did not introduce bias but had the advantage that each successive land cover cell had greater than or equal area available for sites.

Each agriculture and forest cell started with its total area available for construction sites. Each time a site was assigned to a land cover cell, the cells's available area was reduced by the site are. In one HUC (4090001), a point was reached where no land cover cell had enough area to contain the entire construction site. In that case, the code distributed portions of the site onto two different land cover cells. After distributing the sites to land cover cells, quality assurance measures were taken to ensure that:

- The total number of sites distributed in each HUC was equal to the starting value.

- The number of sites in each size category that were distributed in each HUC was equal to the starting values.

- The total site area distributed was equal to the starting site area.
- No land cover cell was assigned more construction site area than was available in the land cover cell.
- The number of land cover cells with sites was close to (or equal, in most cases) to the number of sites in the HUC.

Once construction sites were distributed to the land cover cells, TSS loads were distributed using the HUC, site size, and fraction of site assigned to the land cover cells. The loading file contained total TSS loadings by HUC and site size category, so loadings for each site were calculated by dividing the total load in the size category by the number of sites in that size category. The TSS load distribution process involved several quality assurance measures to ensure that:

- Total TSS loads distributed matched the total loads shown in Table 8-23.
- Total TSS loads within each HUC were the same as in the load file.
- TSS loads by HUC, site size category, and regulatory option were the same as in the load file.

The output of the computer module was a table with the format shown in Table 8-24.

Table 8-24. Example of Output from Site and Load Distribution Process

HUC8	Cell ID	RF3RCHID	Site Size	Fraction of Site	Baseline Load	Opt 2/4 Load
3010102	1	3010102 1 0.00	200	1	30.2	28.7

8.8.2.3 Removal of Background NPS TSS Loads

For each land cover cell that was assigned a construction site, a portion of its background NPS TSS was removed to avoid double-counting. The NPS TSS load on each cell was reduced by the fraction of the land cover cell occupied by construction sites. For example, if a land cover cell was originally assigned 100 ton/yr of TSS, but was assigned a 200 acre (0.81 km^2) construction site, the new NPS TSS load for that cell was calculated as 100 ton/yr * (1-0.81) = 19 ton/yr. This removal process had a negligible impact on NPS TSS loads. Originally, total NPS TSS loads were 5.226×10^8 ton/yr. Approximately 7.126×10^5 ton/yr were removed through this process, leaving a total NPS TSS load of 5.218×10^8 ton/yr. The modified NPS loads underwent an overland transport module that delivered the loads to the RF3 network, and an in-stream delivery module that routed the loads down to the RF3Lite network. For both modeling components, TSS settling was modeled using a net settling velocity approach, as shown in Equation 1.

$$k_{sed} = \frac{v_{sed}}{depth}$$
$$C(x_2) = C(x_1)e^{-k_{sed}t} \qquad (1)$$

where

k_{sed}	=	First-order TSS settling rate (1/day)
v_{sed}	=	Net settling velocity (0.3 m/d; Chapra 1997)
depth	=	Channel depth (m)
$C(x_2)$	=	TSS concentration at x_2 (mg/L)
$C(x_1)$	=	TSS concentration at x_1 (mg/L)
t	=	Time-of-travel from x_1 to x_2 (d)

Table 8-25 presents a summary of these modified NPS TSS loads.

Table 8-25. NPS TSS Loads Modified for Construction and Development Analysis

Scale	TSS Load (ton/yr)	Delivery Ratio
Land Cover Cell	5.22×10^8	N/A
RF3 Network	3.24×10^8	0.62
RF3Lite Network	1.99×10^8	0.38

8.8.2.4 Routing Construction and Development Loads to the RF3Lite Network

The overland transport step was eliminated, which is the same as assuming that all loads from land cover cells entered the RF3 network. This assumption was made because the load development process accounted for the loss of large particles. Construction loads were routed from the RF3 network to the RF3Lite network using the first-order loss approach described in Equation 1. Table 8-26 summarizes the delivery of construction and development TSS loads to the RF3Lite network. TSS loads from construction sites accounted for approximately 1% of the total TSS loads entering the RF3Lite network.

Table 8-26. Summary of Construction and Development TSS Loads

Option	LCC Load (ton/yr)	RF3 Load (ton/yr)	RF3 Delivery Ratio	RF3Lite Load (ton/yr)	RF3Lite Delivery Ratio
Baseline	6,288,751	6,288,751	100%	3,806,800	61%
2/4	5,347,643	5,347,643	100%	3,238,926	61%

8.8.2.5 Water Quality Modeling and Economic Benefits Analysis

After the construction and development and modified background NPS loads were routed to the RF3Lite network, the next step in each model run consisted of water quality modeling in the RF3Lite network using Eutro-WASP and the mean annual flow condition. After in-stream modeling with Eutro-WASP, the WQI6 and WQL values were calculated in each RF3Lite reach. Economic benefits associated with the regulatory options were calculated for RF3Lite reaches that showed a change in WQI6 or WQL.

8.9 RESULTS OF CONSTRUCTION AND DEVELOPMENT MODELING ANALYSIS

Table 8-27 lists the number of improved reaches and the length of the improved reaches for Option 2/4 over baseline conditions. Option 2/4 loads also caused water quality degradation in a number of reaches. This degradation was likely due to effects of algal growth on modeled TSS concentrations.

Tables 8-28 and 8-29 list the economic benefits estimates using both the WQL approach and the WQI6 approach, respectively. The sum of local and nonlocal annual benefits for Option 2/4 ranged from $15,203,000 to $28,357,000 (year $2002). EPA was not able to ascribe any benefits to Option 1.

Table 8-27. Summary of Waters Affected (Option 2/4)

Method	Number of Improved Reaches	Improved Segment Length (miles)	Number of Degraded Reaches	Degraded Segment Length (miles)
WQI6	7,446	9,303	38	78
WQL	583	803.3	26	55.8

Table 8-28. Economic Benefits Using the WQL Approach (Option 2/4)

Use Support Category	WQL Benefit (2002$)*
Boat	$8,461,000
Fish	$15,580,000
Swim	$4,316,000
Total	**$28,357,000**

* Note: numbers may not add due to rounding

Table 8-29. Economic Benefits Using the WQI Approach (Option 2/4)

WQI Category	WQI6 Benefit (2002$)*
WQI<26	$27,000
26 < WQI < 70	$7,714,000
WQI > 70	$7,462,000
Total	**$15,203,000**

*Numbers may not add due to rounding

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